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# Summary

In developing a thematic strategy on the prevention and recycling of waste, the European Commission envisages an important role for the use of economic instruments, such as landfill taxes. The Netherlands is one of the EU Member States applying a landfill tax. The present study aims at investigating the effectiveness of the landfill tax as a waste policy instrument, in terms of market impact, costs and benefits (including external environmental costs and benefits), and to compare it with other (command-and-control type) instruments. The results should contribute to the European debate on landfill taxation.

The study addresses three central questions:

1. To what extent is the Dutch landfill tax providing the right incentives to the appropriate market parties, and is the waste market functioning sufficiently in order to achieve the policy objectives by means of market incentives?
2. To what extent does the landfill tax cover the external costs of landfilling?
3. Is the landfill tax the least expensive option to minimize landfilling, or are there other ('command-and-control type') instruments that are more cost-effective?

In answering these questions, a number of different approaches were followed.

A short literature survey was done on the existing theoretical and empirical insights in the effectiveness of landfill taxes, including experiences in several EU countries. This survey shows that if landfill taxes are to be effective in terms of reducing the amount (and the share) of (municipal) waste going to landfills, the tax rates should be put at a fairly high level. Moreover, the price signals that they convey should be transferred to the sources of the waste. This can be done by introducing unit-based pricing systems for waste disposal services. However, such systems also have potential disadvantages (e.g. high transaction costs, illegal disposal of waste). A careful design of the instrument is therefore needed.

There is also a large variety of estimates for the price sensitivity of waste supply. In some cases, relatively large reductions in waste supply have been recorded following the introduction of unit-based pricing schemes, but these may be partly attributable to an increase in illegal waste dumping. On the other hand, part of the recycling behaviour observed in households seems to be unrelated to the cost of waste disposal and may be better explained by attitude factors.

The literature furthermore suggests that embedding the landfill tax in a mix of policy instruments that promote prevention and recycling can enhance its effectiveness. Differentiating the tax rates (taking account of the environmental features of the landfill) may be effective in speeding up the modernisation of landfills.

Despite these findings, scientific research on the effectiveness of landfill taxes is still scarce. An exploration at European level of the instrument 'landfill tax' would definitely benefit from more research into conditions for effective landfill taxes. The present study is a first attempt to do so for the Dutch situation. The focus is on household waste and comparable waste from the service sector (offices, shops etc.), as these are the waste

streams to which EU policy pays a lot of attention and on which a lot of information is available.

The Dutch landfill tax was introduced in 1995 with the primary aim to increase the financial attractiveness of alternatives to landfilling (i.e. recycling and incineration). The present tax rate amounts to almost € 85 per tonne, which is the highest landfill tax rate in the EU. In 1995, 35% of household waste and 43% of service sector waste was landfilled. By 2003, these percentages had decreased to 6% and 11%, respectively.

An *ex post* analysis was made of the impact of the Dutch landfill tax on the amount of waste and on the choice of treatment option, using statistical (regression) analyses. By the year 2000, when the tax rate was increased substantially, the landfill tax had made landfilling a more expensive waste treatment option than incineration. Whether or not landfilling has also become more expensive than recycling is less clear. It appears that the landfill tax did not have a significant direct impact on the generation of household waste, nor did it affect the choice for household waste disposal options. However, there may be an indirect effect if municipalities pass on the higher costs of landfilling to households by means of a unit-based charge (instead of a 'flat fee') on household waste (in 2004 29% of Dutch municipalities applied such a differentiation). The analysis shows that provinces with a high share of municipalities using unit-based pricing schemes have lower amounts of waste per capita, a lower share of waste landfilled, and a higher share of waste incinerated. Whether there is any causal relationship is unclear, as the choice between landfilling and incineration is not made by the households themselves, but by the municipalities.

For the service sector, the level of disposal costs is not affecting the generation of waste, but it does influence the waste disposal choice. In particular, higher costs for landfilling and incineration increase the share of recycling. Moreover, if the relative increase of costs of landfilling exceeds the relative increase of costs of incineration, firms from the service sector will landfill less waste and incinerate more. In this sense, the landfill tax can play a crucial role in the decision making of firms from the service sector with respect to disposing waste

The higher costs of landfilling due to higher landfill tax levels increase the demand of the service sector for the incineration of waste. Due to a constant capacity and a slightly growing incineration of waste, the efficiency of use of waste incineration plants shows a slightly positive trend. In the case of recycling, we observed a moderate growth in the case of household waste, and a strong growth in the case of the service sector waste. We were not able to link these findings to the efficiency of the use of recycling infrastructure.

During the observed period (1995-2003), no new incineration capacity has come into operation in the Netherlands (though some new plants are presently under construction). However, export of waste for incineration and use as a secondary fuel has increased. So, the landfill tax in the Netherlands might have contributed to a more efficient use of foreign waste incineration capacity and/or new foreign incineration capacity. More research into developments of the European waste market is needed to come to a reliable conclusion on this issue. If the landfill tax has led to investments in recycling infrastructure is ambiguous, because different kinds of recycling options require their own infrastructure.

In future research, the development of recycling options could be investigated in more detail.

The *ex ante* effectiveness of landfill taxes under a number of scenarios was assessed by means of a general equilibrium model. The results of the ‘benchmark scenario’ show that a landfill tax has a significant effect on the amount of waste landfilled. The higher the landfill tax the more waste will be recycled or incinerated. The model predicts that municipalities will start to incinerate all their waste if the landfill tax becomes too high. Only in municipalities that charge a unit-based price for waste collection will households directly notice the effects of the landfill tax by an increase in the price for waste collection and thus start to recycle more waste. In municipalities that charge a flat fee for waste collection, households will not have an incentive to recycle more waste. Recycling efforts, however, are low regardless of the pricing system for waste collection. The increase in the landfill tax will only provide a small price incentive to recycle. Most of the municipal solid waste is already incinerated so the price increase of waste collection due to the landfill tax will be slight.

The service sectors, in contrast to the municipalities, choose (according to the model calculations) to recycle more waste. Some sectors slightly increase their demand for waste incineration services, but the biggest difference in the service sectors is the amount of waste that is recycled. This is mostly caused by the fact that, similar to the Dutch situation, the incineration capacity in the model is too low to accept both an increased amount of municipal solid waste and an increased amount of service waste. If export of combustible waste is allowed, then the service sector will also increase the amount of waste they incinerate. Export of combustible waste will slightly reduce the recycling effort of the service sector.

The model shows that while export of combustible waste will only stimulate producers to incinerate more waste, export of waste to be landfilled has more far reaching effects. If the price of landfilling (including the landfill tax) in the Netherlands exceeds the price of landfilling in the neighbouring countries, the model calculates that all landfill waste will be exported. As a consequence, producers will no longer have a price incentive to recycle or incinerate waste. Thus the landfill tax will no longer be effective in terms of reducing landfilling.

An increase in the landfill tax will decrease the welfare of society, measured in terms of consumption. The impact analysis shows a relatively large decrease in waste landfilled combined with a relatively low decrease in welfare if the landfill tax is increased from a low level to a slightly higher level. If the landfill tax is higher (for example higher than 100 euro per tonne) the decrease of waste landfilled is much lower compared to the loss of welfare.

A landfill ban can also be used to reduce the amount of waste landfilled. The model shows that compared to the landfill tax, the ban is not nearly as cost-effective. To reduce landfilling to zero may involve higher costs than would be socially optimal. Besides, the optimal level of landfilling may not be equal to zero. A landfill tax can be used to reach the optimal amount of landfilling. The landfill ban strictly enforced will not be able to reach this optimal level of landfilling (even though transaction costs may be lower).

The analysis suggests a number of important conditions for the functioning of the waste market in order to make a landfill tax effective. There should be no restrictions on the availability of incineration capacity (both within The Netherlands and abroad). Allowing export of combustible waste increases the effectiveness of the landfill tax, assuming that there is sufficient capacity available in neighbouring countries. Unit-based pricing of household waste collection also enhances the effectiveness of the landfill tax: households will not increase their recycling efforts if they do not 'feel' the higher cost of landfilling. However, the costs of such systems should be weighed against the benefits in terms of the (limited) increases in recycling effort that can be expected. Finally, allowing waste export for landfilling abroad will reduce the effectiveness of the landfill tax significantly (although the amount of waste landfilled in the Netherlands decreases, the amount landfilled abroad increases).

At the present landfill tax rate of almost € 85 per tonne it is already attractive for a lot of waste suppliers to turn to alternatives. However, due to other restrictions (such as a lack of incineration capacity or a ban on the export of waste) they may be forced to landfill their waste anyway. In the absence of such restrictions, higher levels of the landfill tax rate would lead to less landfilling, but at the expense of relatively high social costs. Our analysis does not allow us to draw conclusions on the impact of a differentiation of the tax rate by type of waste, treatment option or waste supplier. However, economic theory tells us that any differentiation should only be based upon differences in external costs of the waste and the treatment option, and not on the type of waste supplier.

The external costs and benefits of landfilling as well as incineration were estimated, using various techniques for the valuation of the environmental impacts (or 'externalities') of both types of waste management. Generally, information from existing international sources was used and adapted to the Dutch situation. In particular, the size of the population exposed to the local impacts of landfills and incineration plants (such as disamenity and health impacts) was calculated using GIS analysis.

Table S.1 shows the estimates of the external, private and net social costs for the waste treatment options landfilling, incineration and co-incineration. 'Negative' costs of displaced energy production are taken into account.

*Table S.1 External, private and net social costs of landfilling, incineration and co-incineration (€/ton, rounded figures).*

| | Landfilling | Incineration | Co-incineration |
|---|---|---|---|
| External costs | 10 | 18 | 28 |
| *Range* | *7 - 79* | *12 – 25* | |
| External costs of displaced energy | -1 | -8 | -4 |
| *Range* | *-1 - -4* | *-7 - -13* | |
| Net external costs | 9 | 11 | 23 |
| *Range* | *6 - 75* | *5 – 13* | |
| Private costs | 36 | 101 | 101 |
| **Net social costs** | **45** | **112** | **124** |
| *Range* | *42 – 111* | *106 – 114* | |

In the case of landfilling, greenhouse gas emissions (mainly methane, which is only partly recovered and used for energy production) and disamenity account for the largest share of the external costs. Due to substantial uncertainties, the range between the 'low' and 'high' estimate is relatively large. The uncertainty range in the external costs of incineration is smaller than in the case of landfilling. Co-incineration leads to higher external costs than incineration, because most emissions per tonne of waste are higher. For recycling, no estimates of net social costs could be made, as many different waste streams are involved and both the external and the private costs can be either positive or negative.

The calculations made strongly suggest that the current landfill tax rate of almost € 85 per tonne is at least equal to, but probably substantially higher than the marginal external costs of landfilling. The 'best' estimate of the external costs is about € 9 per tonne, and even the 'high' estimate (€ 75 per tonne) is lower than the current tax rate.

A definitive conclusion on the 'lowest social cost' option for waste management cannot be given, due to lacking and sometimes contradictory information, especially regarding co-incineration and various recycling options. Nevertheless, it is clear that the net social costs of landfilling are probably much lower than those of (co-)incineration. Some combination of landfilling (with methane recovery) and recycling might well be the strategy implying minimum net social costs. Obviously, such a strategy would be dependent on the availability of sufficient space for (new) landfill sites.

Compared to other instruments, such as a ban on landfilling or an obligation to accept waste, the social costs of a landfill tax are relatively low. In economic terms, a landfill ban is similar to a prohibitively high landfill tax rate. The social costs of reducing the amount of landfilled waste to zero would be very high. A legal obligation to accept waste would also imply significant welfare losses. A well-designed system of environmental taxes will minimize the total social costs of waste treatment, provided that there are no market distortions. This means that waste suppliers should have the opportunity to choose between alternative treatment options (domestically or abroad) and that the tax rates reflect the external costs of the treatment option. In the current Dutch situation, this would probably mean a reduction of the landfill tax rate and an increase in the waste tax rate for incineration (which at present has a zero rate). Both rates should be around € 10 per tonne of waste. However, it is clear that a waste tax of this magnitude would reduce the incentive to divert waste from landfilling to recycling and incineration substantially. Alternatively, tradable landfill permits could be considered. This instrument, which combines the efficiency advantages of a tax with the certainty of a cap on the total amount of landfilled waste, deserves further investigation.

# 1. Introduction

In its Communication 'Towards a thematic strategy on the prevention and recycling of waste' (COM(2003)301) the European Commission states: "The main obstacle to further recycling is the latter's cost disadvantage compared to other waste treatment options. The use of economic and market-based instruments is therefore considered to be the most promising way to promote recycling" (Section 5.3). One of the instruments the Commission deals with in this respect is the landfill tax. On this instrument, the Communication states: "The role of landfill taxes should be explored in the context of this thematic strategy, despite the political sensitivity associated with fiscal measures in general. This would not necessarily imply the introduction of a harmonised Community landfill tax. Closer co-ordination between competent authorities in Member States could be a useful first step to address this issue. This could initially focus on building consensus concerning the effectiveness of landfill taxes and later develop criteria for closer alignment of taxes adopted at national level." (Section 5.3.1).

The Netherlands is one of the EU Member States applying a landfill tax. It was introduced in 1995 and was primarily aimed at bridging the gap between the costs of landfilling and incineration. As such, the choice for this instrument and the rates that are applied in The Netherlands are mainly based on political and administrative considerations and decisions. The present study aims at investigating the effectiveness of the landfill tax as a waste policy instrument, in terms of market impact, costs and benefits, including external environmental costs and benefits, and to compare it with other (command-and-control type) instruments. The results should contribute to the European debate on landfill taxation. The focus is on household waste and comparable waste from the service sector (offices, shops etc.), as these are the waste streams to which EU policy pays a lot of attention and on which a lot of information is available.

The study addresses three central questions:

1. To what extent is the Dutch landfill tax providing the right incentives to the appropriate market parties, and is the waste market functioning sufficiently in order to achieve the policy objectives by means of market incentives?
2. To what extent does the landfill tax cover the external costs of landfilling?
3. Is the landfill tax the least expensive option to minimize landfilling, or are there other ('command-and-control type') instruments that are more cost-effective?

More specifically, a number of sub-questions were formulated:

1a) Has landfilling become a relatively more expensive waste treatment option compared to alternatives such as incineration and recycling[1]?

1b) Has the landfill tax led to a relative increase in waste supply for incineration and recycling, and to a relative decrease in waste supply for landfilling?

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[1] The term 'recycling' is used in a broad sense in this report, and covers all kinds of operations in which waste is being put to useful purposes.

1c) Has the landfill tax led to a better utilisation of the existing infrastructure for incineration and recycling, both in The Netherlands and abroad?

1d) Has the landfill tax led to investments in new capacity for incineration, separation and recycling, both in The Netherlands and abroad?

1e) What are the conditions that the waste market has to fulfill in order to apply the landfill tax as an effective instrument?

1f) What is the desired or optimal rate of the waste tax in order to make the suppliers of waste choose alternative waste treatment options (incineration and recycling)? In particular, should the rate be differentiated according to certain aspects such as type of waste, treatment option and waste supplier?

2a) What are the social costs and benefits of the following treatment options for household waste and comparable waste from firms:

- Landfilling all waste;
- The present situation;
- Terminating the landfilling of combustible waste by shifting it to waste incineration;
- Terminating the landfilling of combustible waste by shifting it to co-incineration (in power plants);
- Terminating the landfilling of combustible waste by optimizing/maximizing recycling.

2b) To what extent can the current level of the Dutch landfill tax rate be regarded as an internalisation of the environmental costs of landfilling?

2c) From a social cost-benefit perspective, what is the optimum way of waste treatment for household waste and comparable waste from firms?

3a) What are the financial consequences (in the short and the long term) of landfill taxation, landfill bans and legal obligations (for landfills and incineration plants) to accept waste, both for the waste suppliers and the waste treatment companies?

3b) Which instrument will (in the short and the long term) lead to the lowest costs of waste treatment for the waste suppliers?

In answering these questions, a number of different approaches were followed. A short literature survey was done on the existing theoretical and empirical insights in the effectiveness of landfill taxes, including experiences in several EU countries (Chapter 2). In retrospective (*ex post*) the impact of the Dutch landfill tax on the amount of waste[2] was analysed using statistical (regression) analyses (Chapter 3). The *ex ante* effectiveness of landfill taxes was assessed (under a number of scenarios) by means of a general equilibrium model (Chapter 4). Furthermore, the external costs and benefits of landfilling as well as incineration were estimated and compared with the 'internal' costs and benefits and with the existing landfill tax rate (Chapter 5). In Chapter 6 of this report, conclusions on the research questions are drawn, based upon the analyses in the preceding chapters.

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[2] As indicated above, the focus of the research was on household waste and comparable waste from offices and the like.

# 2. Landfill tax effectiveness: evidence from literature

## 2.1 Introduction

In many countries in the world, the waste management hierarchy (see Figure 2.1) has been taken as a key element in waste management policy. Especially in Europe, the hierarchy is widely applied as a guiding principle. The hierarchy is based on environmental principles, and implies that waste, depending on its characteristics, should be handled by different methods: a certain amount should be prevented by either reducing the content of waste or by reusing the waste, another share of the waste stream needs to be converted into secondary raw materials, some parts can be composted or used as source of energy, and the remaining may be landfilled.[3]

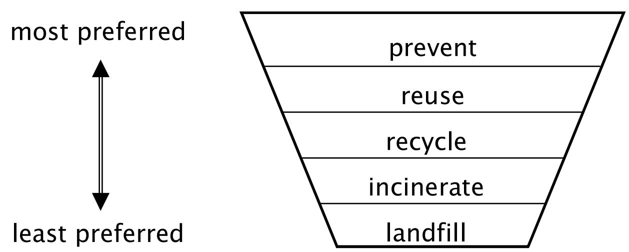


*Figure 2.1 The waste management hierarchy.*

A large set of policy instruments is available to promote the waste management hierarchy. One of them is landfill taxation. Apart from raising revenues (for the general budget or for specific waste related purposes) landfill taxes can provide incentives to divert waste from landfilling towards destinations ranking higher in the waste management hierarchy. Obviously, this incentive will only be effective if the 'waste market' functions well. This means that the price signal from the landfill tax should be transferred to all other elements in the waste chain, and there should be no institutional, regulatory or other barriers impeding actors to respond adequately to the price signal.

This chapter addresses the question to what extent empirical evidence can be found in the literature for the effectiveness of landfill taxation as an instrument to move waste up in the waste management hierarchy.

The chapter is structured as follows. In Section 2.2, the issue of price sensitivity of waste supply is addressed, with an emphasis on household waste. Section 2.3 looks at the empirical evidence with respect to landfill taxes in a number of EU countries. Finally in Section 2.4 some conclusions are drawn.

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[3] It should be realised that the hierarchy has always been subject to fierce criticism. For example, many believe that the options presented in the hierarchy should not be ranked in a particular order but considered as a 'menu' of alternatives. 'It is not a question of good and bad waste management options. Rather, each option was equally appropriate under the right set of conditions addressing the right set of waste stream components' (Schall, 1995).

## 2.2 Price responsiveness of waste supply

### 2.2.1 Advantages and drawbacks of unit based pricing systems

If a landfill tax is to be effective in terms of providing an incentive to produce less landfilled waste and to recycle more, it has to be 'translated' into an incentive for the producers of the waste (in our case primarily households). This means that, ideally, a household should be confronted with the exact marginal costs of each kilogramme of waste it supplies to the waste collection service. Such 'unit based' pricing systems have several advantages, but they also have drawbacks.

One advantage of the unit-based pricing system is that it is directly based on the 'polluter pays principle' as established in the framework directive on waste of the European Union, which, among other things, rules that the cost of waste disposal should be borne by the individual who generates it. The 'polluter pays principle' is generally accepted as instrument of justice given that it not only charges the polluter for the administrative and environmental costs generated by their behaviour, but it also encourages the polluter to mend his ways (Perman *et al.*, 1996). Goddard (1995), however, raises an interesting question regarding the 'polluter pays principle' namely, who is the actual polluter in this case? Is it the consumer who generates the waste by consuming the product, is it the producer who designs a product that contains either too much waste or is not recyclable, is it the package designer, who pays little attention to the waste content of his design, or is it the retailer who desires packaging that keep transaction costs low? It is impossible to answer this question. Goddard demonstrates that it is more appropriate to consider which of the actors is in the best position to control the waste flow. A well-informed consumer would be the proper person to make personal consumption choices. By getting the prices of waste disposal right, the consumers can decide on their own how much municipal solid waste should be prevented or recycled.

Another advantage of the unit-based price is that it ensures an efficient allocation of resources without requiring other tax and subsidy instruments (Fullerton and Wu, 1998 and Palmer and Walls, 1994). If a unit-based price is introduced, households will start to consume, recycle, and dispose waste in such a way that the marginal benefits of consumption and recycling are equal to the marginal costs of disposal. In such a case, the market will provide the proper prices for consumer goods, recycling and disposal. For example, if consumers start to recycle more waste, recycled material becomes cheaper. Thus producers will start to use more recycled material without needing an extra incentive of the government in terms of a recycling subsidy (Kinnaman and Fullerton, 1999). In fact, Dinan (1993) shows that introducing both a unit-based price on waste disposal and a subsidy on the use of recycled material is inefficient as this basically subsidizes the use of recycled material twice.

Several studies, however, have illustrated that the introduction of a unit-base price will lead to significant transaction costs, thus it may be inefficient to introduce such a pricing system. First of all, the administrative costs of introducing a unit-based price may exceed the social benefits of lower waste generation. Fullerton and Kinnaman (1996) estimate that the administrative costs of introducing a unit-based price on the bases of an 'expensive bag' in Charlottesville, Virginia could exceed the $3 per person social benefits mentioned before. Linderhof *et al.* (2001), however, reveal that in Oostzaan the cost of waste collection and disposal did not increase after the introduction of a weight-based price for waste collection. Furthermore, they show that the costs invested in the introduction of the weight-based pricing system are compensated by the lower cost of waste treatment due to the reduction of waste. These results depend largely on the individual municipality. In the case of Linderhof *et al.*, the average consumer in the municipality was very environmentally friendly oriented. Thus, consumers were more than willing to recycle and prevent waste. It can be expected that results in other municipalities would be less positive.

Secondly, Dinan (1993) showed that a uniform unit-based price for all types of waste might be inefficient if materials within the waste stream led to different social costs. For example, the treatment of hazardous waste, such as flashlight batteries, will generate far greater social costs than the treatment of recyclable waste, such as old newspapers. The unit-based price collection of flashlight batteries should, therefore, be higher than the unit-based price for collection of old newspapers. Other studies, such as Walls and Palmer (2001), Eichner and Pethig (2001), and Calcott and Walls (2002), support these results. A solution would be a selective unit-based pricing system based on the social costs of disposing the material in question, but this would of course be rather expensive to implement.

Thirdly and most seriously, the unit-based pricing system may promote the illegal disposal of waste. Households may start to dump their waste in their neighbours' bins, dispose of it at work, illegally dump waste, or burn it themselves. Such behaviour leads to large social costs and has been identified as one of the most serious obstacles to the introduction of a unit-based pricing for waste collection. Both Dobbs (1991) and Fullerton and Kinnaman (1995) demonstrate that if illegal disposal is a possibility, it may be optimal to have a negative tax on waste disposal, *i.e.* legal waste disposal should be subsidized. In such a case, policy makers would be better off implementing other policy instruments to reduce waste generation. Fullerton and Kinnaman (1996) estimate that about 28% of the decrease in waste generation may be caused by increased illegal disposal. Empirical studies, like Jenkins (1993) and Miranda and Aldy (1998), also report instances of increased illegal dumping. These results, however, are contradicted by other empirical studies. For example, Miranda *et al.* (1994), Strathman *et al.* (1995), Nestor and Podolsky (1998), Podolsky and Spiegel (1998), Sterner and Bartelings (1999) and Linderhof *et al.* (2001) found no significant evidence of increased illegal disposal.

Despite the three disadvantages mentioned above, the unit-based price is one of the most effective policy options to provide an incentive to increase prevention and home composting. None of the other policy tools can significantly influence the consumers' choice to prevent waste. Therefore, Calcott and Walls (2002) find that a modest disposal charge will always be part of the set of optimal policy instruments. Shinkuma (2003) even goes a little farther, arguing that even if illegal disposal is an option, the unit-based pricing system will still provide a second best optimum as long as the price of recycled material is positive. Only if the price of recycled material is negative, should another policy tool like the deposit-refund system be considered.

### 2.2.2 Elasticities of demand for waste disposal services

The success of market based policy instruments depends on the elasticity of demand[4] for waste disposal services. For example, a unit-based price for waste disposal will only affect the disposal of waste if the demand for disposal services is sensitive to the price of the waste disposal services. As municipalities have been experimenting with the introduction of recycling programs, unit-based pricing, and deposit/refund systems, a large range of empirical studies discussing the price elasticity of waste generation have been conducted. In this Chapter, we will discuss the most important literature on this subject.

Wertz (1976) analyzed the households' responsiveness to unit-based prices. By comparing the average quantity of waste generated in San Francisco, a city with a user fee, with the average quantity generated by an average town of the United States, without a user fee, Wertz calculated a price elasticity of demand equal to –0.15.

Hong *et al.* (1993) examined the effects of volume-based pricing using a survey of 2298 households from Portland OR, USA. Hong *et al*. estimated a price elasticity of demand equal to –0.03 and an income elasticity of 0.049 suggesting that unit-based pricing only affects demand in a minimal way. They did, however, find that the demand for recycling services is influenced positively by the introduction of volume-based pricing. They also concluded that households are less likely to increase recycling if recycling requires more effort and that a larger household is not only more likely to recycle, but also to generate more waste than a smaller household.

Jenkins (1993) gathered data from 14 municipalities in the United States (including 10 municipalities that charged a unit-based price) over several years. She found an inelastic demand for waste disposal, reporting a price elasticity of –0.12. Jenkins concluded that waste generation and recycling are positively influenced by the size of the household. However, she also found the effect to be statistically insignificant.

Miranda *et al.* (1994) used data from a 21-city sample to estimate the effects of introducing a unit-based price. They found that unit-based pricing provides residents with a strong incentive to both reduce waste and recycle it. They note, however, that most municipalities implement a unit-based price in combination with an aggressive recycling program. In the one municipality that introduced unit-based pricing on its own, the experiment failed. Households turned to private waste collectors and illegal disposal increased significantly. Therefore, this municipality chose to return to the flat fee-pricing system. This evidence, although anecdotal, seems to suggest that a unit-based pricing scheme cannot be successful without a recycling program.

Morris and Holthausen (1994) use a household production model to simulate responses to different pricing systems using calibration techniques. They estimate that the elasticity of demand for waste disposal services was in the range between –0.51 and –0.6.

Reschovsky and Stone (1994) employed an econometric model to estimate the actual household responses to unit-based pricing. They used data from 3040 households from Tompkins County, New York. Based on these data, they estimated income elasticities for

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[4] The elasticity of demand is the percentage by which the demand for a product or service (in this case a waste disposal service) increases if the price of the product or service increases by 1%. Obviously, the value of the elasticity of demand will normally be negative.

the demand of collection services equal to 0.23 in case of the introduction of volume-based pricing and 0.24 in case of the introduction of weight-based pricing. These results are quite similar to the results found by Wertz (1976). Reschovsky and Stone try to determine how much waste was illegally disposed of. They found that much of the illegal dumping takes place in the form of the use of alternative dumping facilities, such as roadside dumpsters. They were unable to determine how often illegal dumping or burning occurred. They argued that households are not quite as sensitive to the increased marginal costs of waste disposal as they are to the increased marginal costs of waste reduction. Thus households will only try to reduce waste generation if the marginal costs of waste reduction do not increase too greatly. If the marginal costs of waste reduction increase too much, households will dump waste illegally to reduce the costs of waste disposal. These results suggest that households may have an aversion towards the introduction of a unit-based pricing, indicating that municipalities would be wise to combine a unit-based price with recycling programs or subsidies. Introducing a unit-based price without such a program would be unpopular and less effective.

Strathman *et al.* (1995) estimated the price elasticity of demand for solid waste disposal services using data from Portland, Oregon. They used data on the generation of waste during January 1984 to December 1991. They found an elasticity of demand of –0.45. Strathman *et al.* note that they may have overestimated the absolute elasticity as they expect that the propensity of illegal disposal may be somewhat higher in the Portland region due to the large amount of public land available in this area.

Fullerton and Kinnaman (1996) used household data that were not based on self-reported surveys. They gathered data about the weight and volume of municipal solid waste and recycling efforts of 75 households four weeks prior to, and following the introduction of a volume-based price in Charlottesville, VA. In this municipality, a recycling program had already been operational for about a year. They found that the quantity of solid waste generated decreased only slightly, but that the volume of the waste collected decreased all the more. The density of the municipal solid waste increased significantly, from 15 pounds per bag to just over 20 pounds per bag. They estimated that the introduction of the unit-based price resulted in ten percent less waste, four percent more illegal dumping, and 14 percent more recycling.

Callan and Thomas (1997) found that the implementation of a unit-based price would increase the portion of waste recycled by 6.6 percent. If the introduction of a unit-based price was combined with the introduction of a recycling program, the portion of waste recycled would increase by 12.1 percent.

Kinnaman and Fullerton (2000) were the first to estimate both the levels of recycling and the level of waste disposal simultaneously after the introduction of a unit-based price. They estimate that the cross price elasticity of demand[5] for recycling is 0.220. Moreover, they not only found that an implementation of a $1 unit-based price can decrease the quantity of rest waste generated by 415 pounds per person year, but that it would only increase the quantity of recyclable waste by 30 pounds per person per year. The differ-

[5] The cross price elasticity of demand is the percentage by which the demand for a product or service (in this case recycling) increases if the price of another product or service (in this case: waste disposal) increases by 1%.

ence can be partly explained by increased home composting and prevention, but also points towards the increased illegal disposal of waste.

Although the calculated elasticity of the demand for waste disposal services differs quite a lot between different studies, we can conclude that the demand for waste disposal services is inelastic. The introduction of a unit-based price will result in a reduction of waste. At least part of this reduction, however, may be caused by increased illegal disposal of waste. It is difficult to give definite empirical proof of increased illegal disposal. Although survey respondents claim that illegal disposal has increased after the introduction of a unit-based price, municipalities have not reported increased costs due to illegal dumping and littering.

The empirical studies discussed above report various elasticities of demand for waste disposal services. These differences may be partly explained by differences in attitudes of the households. Several empirical studies have analyzed why consumers recycle or compost at home. In the next couple of paragraphs, a brief overview of these studies is given. For a more extended overview, see Fenech (2002).

Several studies, for example, Hornik *et al.* (1995), McDonald and Ball (1998), Callan and Thomas (1999), Bruvoll *et al.* (2000, 2002), Tucker and Speirs (2002), Ando and Gosselin (2003) and Jenkins *et al.* (2003), have shown that the opportunity cost of time is a significant determinant for recycling of materials. The more households have to do to recycle and separate waste, the less willing they are to do so. A majority of the consumers are willing to pay a private company about 20 dollars a year to take away the burden of separating waste (Bruvoll *et al.,* 2002). Jenkins *et al.* (2003) conclude that consumers are more likely to recycle materials like aluminum, or paper given that effort in recycling these materials is less than other materials such as glass, plastic, and organic waste.

Recycling behavior is influenced by socio-economic factors such as income, education, population density, single or multiple family dwellings, household size and average age of the head of the household. Most empirical studies, like Jenkins *et al.* (2003), find that income and education are positively correlated with recycling. Population density is negatively correlated with recycling and specifically with home composting of organic waste. An explanation for this correlation is the growing scarcity of suitable outdoor storage of waste as the population density increases. Age and household size have a positive correlation with recycling. Ando and Gosselin (2003) show that multi-family dwellings are less likely to recycle than single-family dwellings. They find that differences in recycling convenience and household demographics are the main reason why this occurs.

Introducing a unit-based pricing system not only increases recycling of waste, but also changes the attitude of households towards waste. Fullerton and Kinnaman (1996) demonstrate that the introduction of a volume-based pricing system in Charlottesville, USA did not so much decrease the quantity of waste generated, but instead decreased the volume of the waste generated. Households reduced the number of bags they generated by crushing the waste down in size, rather than by preventing or recycling it. Households, however, were already participating in voluntary recycling programs before the introduction of the volume-based price, thus the incremental benefit of the volume-based price was low.

Sterner and Bartelings (1999) show that the introduction of a unit-based price in Tvååker, a municipality in Sweden, led to a significant reduction of the quantity of waste collected and that the quantity of waste recycled increased. With an extensive survey of about 600 households and focusing on the motivation behind recycling, they demonstrate that whilst people are encouraged by economic incentives, this is not the only reason why they start to recycle. The amount of time and effort invested in recycling are greater than can be purely motivated by savings on their waste management bill. Halvorsen and Kipperberg (2003) support this conclusion. Berglund (2003) analyses the effect of moral motives on household recycling. He finds that moral motives significantly reduce the costs associated with recycling efforts, thus consumers are more willing to recycle even when they are not financially compensated for doing so.

## 2.3 Empirical evidence of landfill tax impacts

### 2.3.1 Introduction

Taxes or levies on waste going to landfill have been introduced in several EU member states. The highest rates are found in the Netherlands (€ 84 per tonne for low density waste and € 13 per tonne for non-combustible high density waste), Flanders and Denmark (more than € 50 per tonne), Austria and Sweden (more than € 40 per tonne), Wallonia, UK and Finland (more than € 20 per tonne). Ireland, France, Czech Republic, Italy and recently Cataluña have introduced landfill taxes of € 7-15 per tonne (CEWEP, 2004). Denmark and the Netherlands, two countries that implemented landfill taxes early on and have set those taxes at relatively high levels, also have the lowest dependency on landfill and highest levels of waste recovery (Integrated Skills LTD, 2004). However, there is no straightforward correlation between high landfill taxes and low landfill rates (see Figure 2.1). Other factors, such as available space and regulations concerning the landfilling of waste also play a role, and therefore one should resist the temptation of jumping to conclusions regarding the effectiveness of landfill taxes.

### 2.3.2 The Netherlands

The landfill tax[6] in The Netherlands was introduced in 1995 at a level of NLG 29.20 (€ 13.25) per tonne. Its current rate (2005) is € 84.78 per tonne (a lower rate of € 13.98 applies to certain categories of waste, including hazardous waste and waste with a density of more than 1100 kg per $m^3$).

In the period 1995-2003 the amount of landfilled waste in The Netherlands decreased by around two thirds from 8.215 ktonne to 2.753 ktonne. During the same period, the amount of waste incinerated increased by around 75% from 4.695 ktonne to 8.218 ktonne. The amount of recycled waste increased by around 30% in the same period from 38.435 ktonne to 49.936 ktonne (MNP, 2005, Bijlage 2). These figures are not only the result of the landfill tax: other factors, such as a ban on the landfilling of combustible waste, also play a role. This is further investigated in chapter 3 of the present study.

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[6] Formally, it is a tax on both landfilling and on incineration, but the rate for incineration is zero.

### 2.3.3 UK

The UK landfill tax was introduced in 1996 with the explicit intention of internalising externalities associated with landfill. Evidence on its effectiveness is mixed. An early evaluation (cited in EEA, 2000) suggested a strong increase in recycling activities after the introduction of the tax. Ecotec (2001) concluded that the impacts on municipal waste have been rather limited. There was no slowdown in the growth of household waste generated, and the growth in recycling could be attributed mainly to other factors (such as packaging regulations). On the other hand, the tax almost certainly has had a strong impact on construction and demolition waste. The tax rate for the latter category ('inert' waste) was much lower in absolute terms (GBP 2 per tonne, against GBP 7 to 15 for 'active' waste), but higher in relative terms (increasing the cost of landfilling for inert waste by up to 200%). It was estimated that as much as 36 million tonnes per annum of inert wastes might have been diverted away from landfill in the wake of the tax (Ecotec, 2001, p. 307).

According to Martin and Scott (2003), the landfill tax has had a relatively low impact on the generation and disposal of waste in the UK. Municipal waste continues to grow at rates exceeding economic growth. This is occurring despite the steady increase in taxation burden. Landfill is still relatively cheap in the UK (ACBE, 2004). The main reason for the ineffectiveness has been the incorporation of the landfill tax with other municipal taxes into a flat fee. In terms of encouraging other waste management options, the tax has been too small (Ecotec, 2001).

The assessment of the landfill tax by the British government is more upbeat. In its Budget 2005 (Box 7.4), the government states that the total volume of waste disposed to landfill fell by almost 20 per cent between 1997-98 and 2003-04, and the quantity of inactive or inert waste fell by 60 per cent over the same period. It concludes that the landfill tax has been effective in diverting waste from landfill. In order to further reduce the landfilling of biodegradable municipal waste, the rate of the landfill tax on active waste will increase annually by at least GBP 3 per tonne to a medium- to long-term rate of GBP 35 (€ 52) per tonne.

Undesirable outcomes of the UK landfill tax include an increase in fly-tipping and a diversion of waste to unlicensed sites. There is no doubt that there has been a substantial and additional burden to local authorities as a result of the tax. Of fundamental concern to local authorities is their ability to bring about changes in waste management practice and to influence the pace of change. The Local Government Association amongst others pointed out that "local authorities are often constrained by long term contracts and it will take time for new waste management facilities to become available". The organisation argued that as "householders do not directly bear the costs of the wastes they generate, there are no incentives for individuals to reduce their generation and levels of household waste continue to show an increase in some areas" (House of Commons, 1998).

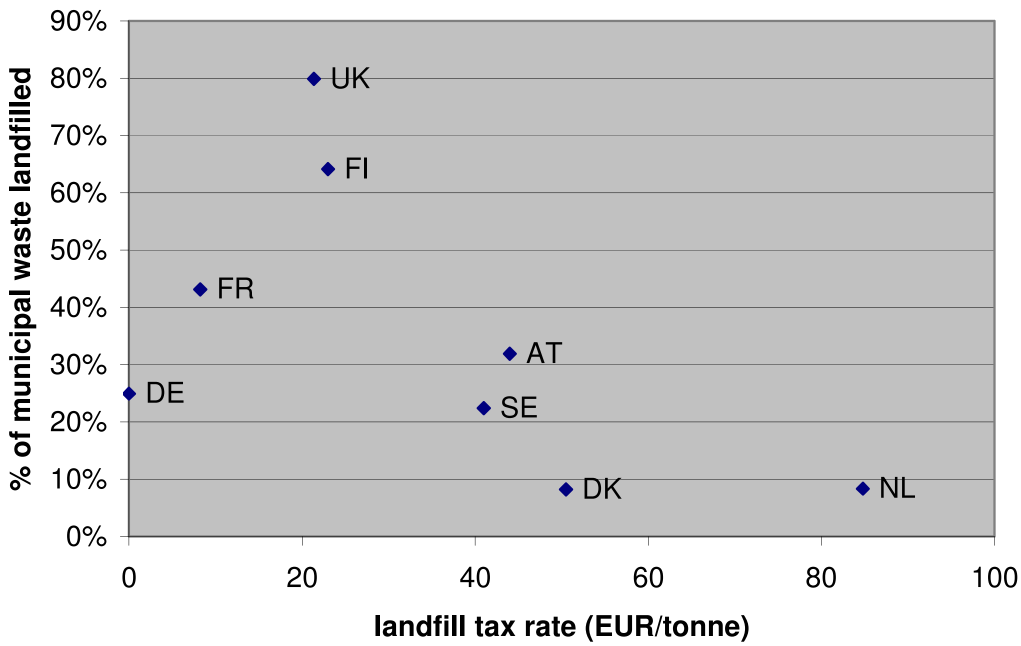


*Figure 2.2. Landfill tax rates and percentage of municipal waste landfilled in selected EU countries (based on data from Eurostat and CEWEP).*

### 2.3.4 Denmark

Denmark introduced a waste tax already in 1987. Initially, the rate for landfilling and incineration was the same, but since 1993 it is differentiated (with a higher rate for landfilling). In 1997 the rates were increased substantially; the rate for landfilling became DKK 335 (about € 45) per tonne.

According to Andersen (1998), in the period 1987-1996 Denmark achieved a 26 percent reduction in the quantity of waste brought to its municipal landfills and incinerators and attained an overall recycling rate of 61 percent. This was the result of a comprehensive waste reduction policy with several elements, including the waste tax. More than 80 percent of the reduction occurred in areas not subject to regulation, such as construction materials and garden waste, where the establishment of new recycling facilities played a prominent role. Particularly in the case of construction materials, the waste tax may have been important in promoting recovery and reuse. Overall, municipal waste authorities were more responsive to the waste tax than were corporate and institutional leaders, which is surprising as the former are in a position to simply pass the tax on to local residents. A possible explanation offered by Andersen is the fact that waste management is the primary focus of municipal authorities, whereas for most firms and institutions it is a side issue that attracts limited attention from top management.

Dengsøe and Andersen (1999) report that after the large increase in waste tax rates in 1997, the amount of taxable waste delivered at municipal waste treatment plants (landfills and incineration plants) decreased by at least 0.5% from 1996 to 1998, despite an economic growth of 7% and despite the fact that more types of waste became taxable. They also noticed that due to the significant differentiation in tax rates between landfilling and incineration, the latter had become a popular alternative to recycling. Householders did not have a financial incentive to increase recycling, because the waste tax was not reflected in the waste collection fees. This lack of transmission of the price signal of the waste tax in financial transactions in the waste sector, especially households, was assessed to constitute one of the decisive barriers to a decrease in taxable waste amounts.

### 2.3.5 Austria

In Austria, the landfill tax was introduced in 1989 with the aim of raising revenues for the clean up of contaminated sites. Since 1996, its rates are differentiated according to the technical quality of the landfill site and to the type of waste. Landfills with state-of-the-art technology pay a much lower rate than sites without any anti-pollution provisions, e.g. against leakage of landfill gas. Since 2004, the standard rate for landfills with state-of-the-art technology is € 21.80 per tonne, whereas for other landfills it is € 65 per tonne (to be increased to € 87 in 2006). According to the Austrian Federal Environment Agency, this differentiation has been a clear incentive to modernise the Austrian landfills: whereas in 1996/97 21 sites did not meet the latest technological standards, in 1999 this was true for only 4 sites (Umwelbundesamt, 2000).[7] On the other hand, there is little evidence to suggest that the tax has had an impact in reducing waste or on the share of waste going to landfills. Between 1989 and 1999 the share of household waste whose final deposit was landfill fell from 75% to 43%, but it is unclear to what extent this can be attributed (if at all) to the landfill tax or more to the various regulations, and awareness meaures to encourage recycling and composting (Ecotec, 2001).

In 2004, the 'Deponieverordnung' introduced stringent restrictions on landfill authorisations. Henceforth, only pre-treated and harmless waste is allowed to be landfilled.

### 2.3.6 Finland

The Finnish tax on waste landfilling was introduced in 1996. Since January 2005, its rate amounts to € 30 per tonne. According to Kautto and Melanen (2004) economic intruments (including the waste tax) have stimulated companies to increase the recovery of waste, although in most cases the costs of waste management were relatively low. The authors argue that this can be at least partly explained by the fact that many of the companies they studied were located in municipalities in which the municipal waste charges had traditionally been low. When the charges had grown relatively rapidly in the late 1990s and the national waste tax had been implemented in 1996, this had given the firms a signal concerning anticipated developments, making them search for new ways to minimise wastes. Many of their interviewees also felt that taxation and other economic instruments are appropriate tools for enhancing eco-efficiency and sound waste management.

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7 In Norway (not an EU Member State) a similar kind of differentiation in landfill tax rates was introduced on 1 July 2003. The rate for landfills that fulfil EU requirements became NOK 327 (€ 39) per tonne and for those that do not it became NOK 427 (€ 51) per tonne (OECD, 2004). Information on the impact of this differentiation is lacking.

Currently, a study is being carried out for the Finnish Ministry of Environment on the effectiveness of the landfill tax.[8]

### 2.3.7 Sweden

In Sweden, a tax on the landfilling of waste was introduced in 2000 at a rate of SEK 250 (about € 27) per tonne. In 2002, the rate was increased to SEK 288 and simultaneously requirements concerning the separation of combustible waste and a ban on dumping separated combustible waste entered into force. In 2003, the tax rate was further increased to SEK 370 (about € 40) per tonne. The aim is to halve the amount of waste landfilled by 2005 from 1994 levels (OECD, 2004). A report by the Swedish Environmental Protection Agency (Naturvårdsverket, 2003) stated that although the amount of waste going to landfill had actually decreased, there was much doubt as to the quantified effects of the landfill tax. This was because the tax was evaluated when it had only been in place for a short time, there was limited access to the data that could be used for the evaluation, and waste management policy includes a number of other instruments alongside the landfill tax. At the time of writing, it was too early to evaluate the effect of the increase in the tax rate of 2003.

### 2.3.8 France

The French landfill tax was introduced in 1993 at a level of FRF 20 per tonne. The rate increased to FRF 60 (€ 9.15) per tonne in 1995, which is still the standard rate for municipal and comparable waste. Since 2003, sites with EMAS or ISO 14000 certification pay a reduced rate of € 7.50 per tonne. Non-authorised landfills pay a rate of € 18.29 per tonne of municipal waste, which is also the rate for special industrial waste.[9] Since 1999, the tax is part of the TGAP (Taxe Générale sur les Activités Polluantes).

Although the tax rate is relatively low and landfilling is often still the cheapest option, the combination of the landfill tax with regulations (including a ban on the landfilling of untreated waste in 2002) has contributed to the fact that the percentage of landfilling in total waste treatment is not increasing (FEAD, 2003). According to Ecotec (2001), the share of landfilling in waste disposal in France has decreased from 63 to 59% over the period 1993-1997. Although municipalities typically have fixed-term contracts with landfill operators, some have switched to incineration and sorting/recycling since 1997.

### 2.3.9 Belgium (Flanders region)

The region of Flanders has a complicated system of landfill (and incineration) taxes. The rates depend on the type of waste and the type of landfill. Relatively low rates (between € 0.32 and € 7.73 per tonne) apply to specific waste from mining and mineral industries, and to recycling and soil sanitation residues. The rate for inert waste and for inert asbestos is € 10.83 per tonne. For free asbestos and fly ash from thermal power plants the rate amounts to € 18.54 per tonne. For municipal waste it is € 58.73 if the landfill gas is used

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8 Personal communications, Mr Timo Parkkinen (Ministry of Environment) and Ms Riita Kojo (Suunnittelukeskus Oy), 07.04.2005.

9 Source: http://www.douane.gouv.fr (accessed 13.04.2005).

for energy production, and € 61.82 otherwise. The rate for landfills operating without a license is € 123.63 per tonne (source: OVAM; rates are for 2005). Since 2000, there is a ban on the landfilling of untreated municipal waste (with some exemptions).

Since a few years, the amount of household waste in Flanders has ceased increasing. In 2002, it decreased by 0.2% and in 2003 by 3.4%. In 2003, 70% of the household waste was collected separately and most of it was re-used, composted or recycled. Major factors behind these figures are the widespread availability of separate collection facilities and the application in almost all Flemish municipalities of some kind of differentiation in waste collection charges. Of the remaining (non-separated) waste, only 16.3% was landfilled (compared to 52.6% in 1991). Both the restrictions on landfilling and the tax (which makes landfilling more expensive than incineration) are held responsible for this decrease (Sanders *et al.*, 2004).

## 2.4 Conclusions

If landfill taxes are to be effective in terms of reducing the amount (and the share) of (municipal) waste going to landfills, the price signals that they convey should be transferred to the source of the waste, i.e. households and firms producing comparable types of waste. This can be done by introducing unit-based pricing systems for waste disposal services. Such systems are in accordance with the ‘polluter pays principle’ and can contribute to an efficient allocation of resources, because they imply a non-zero marginal cost of waste and thus stimulate recycling and waste prevention. However, unit-based pricing systems also have some disadvantages. They sometimes involve high transaction costs, and tend to stimulate the illegal disposal of waste. Factors that seem to improve the effectiveness of unit-based pricing systems are:

- Embedding unit-based pricing systems in a mix of instruments that promotes prevention and recycling;
- Keeping the charges modest;
- Including a distinction between different types of waste;
- Designing the system in a way that rewards recycling and source separation.

In general the conclusion is that experiences with such systems in different countries are mixed.

There is also a large variety of estimates for the price sensitivity of waste supply. The elasticity of demand for waste disposal services is generally found to be somewhere in the range between –0.1 and –0.5. In some cases, relatively large reductions in waste supply have been recorded following the introduction of ‘pay-as-you-throw’ schemes, but these may be partly attributable to an increase in illegal waste dumping. On the other hand, part of the recycling behaviour observed in households seems to be unrelated to the cost of waste disposal and may be better explained by attitude factors.

High landfill taxes are associated with higher levels of waste recovery across the EU (Integrated Skills LTD., 2004) but this may well be related to other policies as well, such as landfill bans on certain waste streams and regulations on recycling. All countries that have introduced high landfill taxes also apply bans or other restrictions on landfilling, which makes isolation of landfill taxes as a factor behind the decrease in landfilling problematic. However, putting it the other way round, embedding a landfill tax in a mix of policy instruments that promote prevention and recycling seems to be an important success factor or at least a best practice. Another factor that influences the effectiveness of landfill taxes is the tax level. Low levels do not seem to be effective (Case UK). Also fixed-term contracts between waste producers an landfill operators may be an obstacle for the effectiveness of a landfill tax (Case France). The Austrian experience suggests that differentiations in landfill tax rates (taking account of the environmental features of the landfill) may be effective in speeding up the modernisation of landfills. In general this short survey of experiences with landfill taxes in several countries shows that scientific research on the effectiveness of landfill taxes is scarce. A final conclusion is therefore that an exploration at European level of the instrument landfill tax as suggested in the Communication 'Towards a thematic strategy on waste prevention and recycling' would definitely benefit from more research into conditions for effective landfill taxes.

# 3. Ex post assessment of the landfill tax in the Netherlands

## 3.1 Introduction

In 1995, the landfill tax was introduced in the Netherlands. In this chapter, we try to evaluate *ex post* the role of the landfill tax in the Dutch waste market. The following research questions (numbered 1a through 1d in Chapter 1) are addressed:

- Has landfilling become a relatively more expensive waste treatment option compared to alternatives such as incineration and recycling?
- Has the landfill tax led to a relative increase in waste supply for incineration and recycling, and to a relative decrease in waste supply for landfilling?
- Has the landfill tax led to a better utilisation of the existing infrastructure for incineration and recycling, both in The Netherlands and abroad?
- Has the landfill tax led to investments in new capacity for incineration, separation and recycling, both in The Netherlands and abroad?

Actually, we limit our scope of the waste market to waste from households and from other economic sectors that have comparable waste flows, such as the service sector.[10]

In this ex-post assessment we try to identify the impact of the landfill tax on different kinds of developments in the waste (disposal) sector, such as waste supply and disposal choice. In fact, the landfill tax is merely an additional cost component of landfilling waste, and the focus of the ex-post assessment is on the costs comparison of disposal options. Moreover, next to the landfill tax, the waste sector has been affected by a number of regulations and developments in the last decades, such as the landfill ban on combustible and recyclable waste, and the ban on export of particular types of waste.

The structure of this chapter is as follows. In Section 3.2, we present a general overview of the waste market in the Netherlands, including waste supply, waste disposal options, waste policy measures, economic and demographic developments, and the total capacity of incinerators and landfill sites. Secondly, we discuss the methodology how we identify the main determinants (including waste disposal costs) of the waste supply and the waste disposal options in Section 3.3. For convenience, the analysis of the impact of the landfill tax on disposal options is divided into two stages, namely recycling versus waste disposal (incineration or landfilling), and incineration versus landfilling of waste. In both cases, the landfill tax is part of the landfilling costs. Section 3.4 discusses the data used in and the results of the analysis applied to household waste and service sector waste. Finally, Section 3.5 concludes.

[10] The service sector includes wholesale and retail market sector, hotel and catering industry, financial and insurance industry, transportation, and non-commercial services, such as public management, health care, education etc.

## 3.2 The Dutch waste market

### 3.2.1 Developments in the waste market

Figure 3.1 shows (on a per capita basis) the developments in the total amount of waste generated in the Netherlands and the Gross Domestic Product (GDP) of the Netherlands in the period 1995-2003. In this period, the total amount of waste generated in the Netherlands increased from 52.8 to 61.8 Mton. Figure 3.1 shows that the total amount of waste per capita has been declining since 2000. As the total amount of waste per capita declined, GDP per capita still grew until 2001.

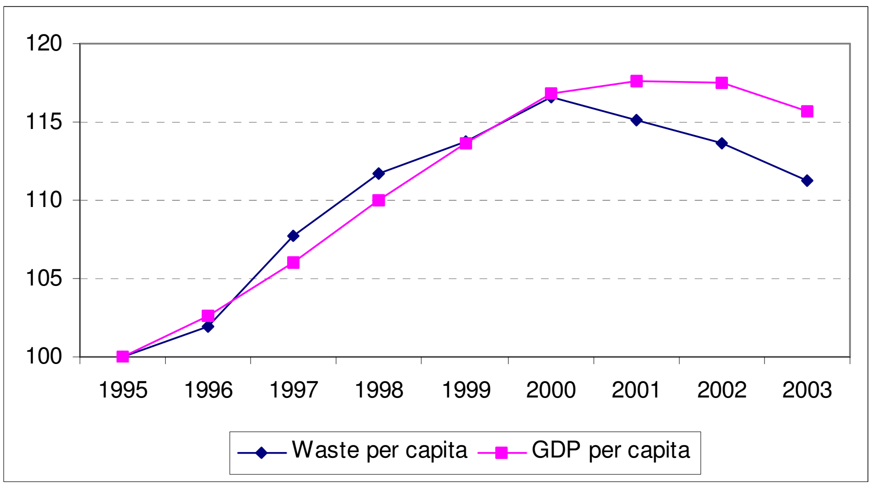


*Figure 3.1 Development of waste and GDP per capita in the Netherlands in the period 1995-2003 (base year is 1995) (Source: AOO and Statistics Netherlands).*

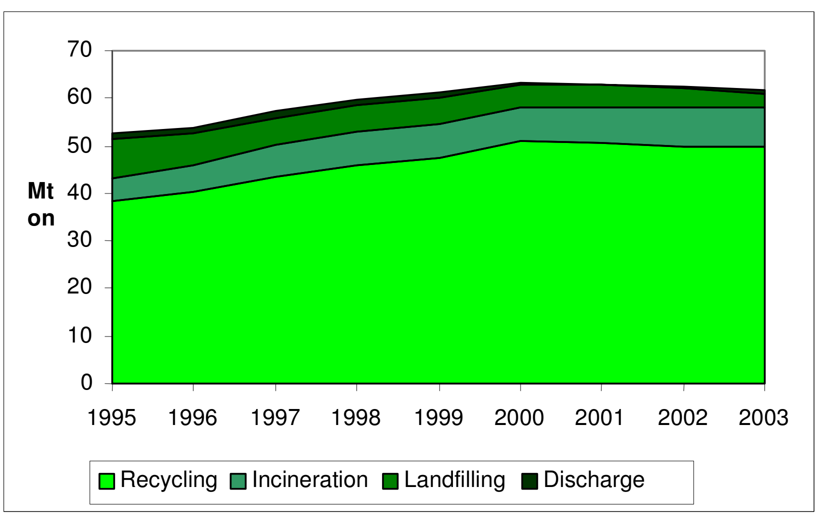


*Figure 3.2 Amount of waste per waste disposal option in the Netherlands in the period 1995-2003 (Source: AOO).*

## Waste disposal options

Figure 3.2 shows the amounts of waste per waste disposal option. Roughly 70-80% of the total amount of waste in the Netherlands is recycled. Here, recycling is used in a broad fashion, because it includes reuse of products, 'pure' recycling of waste and composting organic waste. The total amount of landfilled waste more than halved in the period 1995-2003. In 1995, 8.2 Mton waste was landfilled, while in 2003 the amount of landfilled waste declined to 2.8 Mton. The share of landfilling in the waste disposal options dropped from more than 15% in 1995 to 4.5% in 2003. For incineration, the opposite development is observed. In 1995, 4.8 Mton of waste was incinerated, and the amount increased to 8.2 Mton in 2003. As is mentioned below, however, the domestic incineration capacity in the Netherlands amounts to 5.3 Mton, so that a substantial part of this 8.2 Mton waste has to be exported for incineration.

*Table 3.1 Share of disposal option for household and service sector waste in the Netherlands, 1995-2003.*

| Year | Household waste | | | | Service sector waste | | | |
|---|---|---|---|---|---|---|---|---|
| | Recycling | Incineration | Landfilling | Discharge | Recycling | Incineration | Landfilling | Discharge |
| 1995 | 40.0% | 25.5% | 34.6% | 0.0% | 38.5% | 18.3% | 43.2% | 0.0% |
| 2000 | 44.8% | 40.3% | 14.9% | 0.0% | 55.7% | 17.4% | 26.9% | 0.0% |
| 2001 | 44.5% | 41.7% | 13.8% | 0.0% | 52.6% | 17.8% | 27.5% | 2.2% |
| 2002 | 47.5% | 41.1% | 11.3% | 0.1% | 53.2% | 29.5% | 15.7% | 1.6% |
| 2003 | 51.2% | 41.2% | 5.9% | 1.7% | 53.5% | 33.0% | 11.2% | 2.3% |

Source: own calculations on data from Milieucompendium (http://www.milieucompendium.nl) and MNP (2005).

Since this study particularly focuses on household waste and waste of the service sector, Table 3.1 shows the shares of waste disposal options for those two waste categories. The total amount of household waste increased steadily from 7.3 Mton in 1993 to 9.1 Mton in 2002. In 2003, the amount of household waste declined slightly. Table 3.1 shows that the shares of recycling and incineration have increased between 1995 and 2003. In 2003, more than half of the household waste was recycled (or composted) and 41% was incinerated. The share of landfilling declined from 35% in 1995 to 6% in 2003. The latter figure reflects the exemptions from the general landfill ban on recyclable and combustible waste, which was introduced in 1995. These exemptions are caused by a shortage of incineration capacity in certain regions such as the province of Limburg.

The amount of waste of the service sector increased from 3.1 Mton in 1995 to 3.7 Mton in 2002. In 2003, the amount of service sector waste declined slightly. The share of recycling grew rapidly, and at least since 2000 more than half of all waste generated by the service sector is recycled (or reused). Until 2002, the share of incineration was fairly constant at about 17%, but afterwards it increased to 33%. One explanation for this boost is the increase in exports of disposable (combustible) waste abroad. Simultaneously, the share of landfilling, which was 27% in the period 2000-2001, dropped to 11% in 2003.

## Disposal capacity in the Netherlands

The number of landfills declined from 46 in 1995 to 30 in 2003 (see Table 3.2). Over the same period, the total amount of cumulated landfill capacity decreased from 79 to 51 mln $m^3$. Next to the current landfill capacity, there are still applications for permits to ex-tend existing sites and to open up new sites in progress. The capacity of these permit applications amounts to another 16 mln $m^3$.

In 1995 and 1996 the last incineration plants were built, and from that time the government maintained a moratorium for ‘traditional’ grate incineration plants. New plants using more energy efficient technologies (such as co-incineration or gasification) were allowed, but no new incineration plants became operational until 2003.[11] So, the total number of waste incineration plants was constant at 11 in the period 1997-2003. In this period the annual capacity fluctuated within the range of 5.1 to 5.7 Mton due to annual fluctuations in maintenance activities. The last column of Table 3.2 shows that the efficiency of use of the waste incineration plants (i.e. amount of waste incinerated divided by the capacity) fluctuated between 77% in 1996 and 95% in 2002. There seems to be a positive trend in the efficiency of use of incineration plants, which is best illustrated by the observation that in the period 2002-2003, the efficiency of use has exceeded the level of 90%.

*Table 3.2 Characteristics of waste incineration plants and landfill sites in the Netherlands, 1995-2003.*

| Year | Landfill sites | | | | | Waste incineration plants | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Amount of waste* | | Number of sites | Landfill capacity | | Amount of waste | Number of plants | Capacity | Capacity utilisation |
| | Gross | Net | | Present | In procedure | | | | |
| | Mton | Mton | | mln $m^3$ | mln $m^3$ | Mton | | Mton | % |
| 1995 | 9.9 | | 46 | 79.3 | 28.1 | 2.8 | 7 | 3.4 | 83.9 |
| 1996 | 8.5 | | 47 | 76.3 | 17.1 | 3.5 | 9 | 4.6 | 76.7 |
| 1997 | 7.4 | | 44 | 73.7 | 14.2 | 4.4 | 11 | 5.1 | 86.1 |
| 1998 | 7.1 | | 41 | 69.4 | 6.6 | 4.5 | 11 | 5.3 | 86.3 |
| 1999 | 7.6 | 6.3 | 38 | 63.4 | 6.6 | 4.9 | 11 | 5.4 | 90.0 |
| 2000 | 6.6 | 5.6 | 36 | 56.7 | 17.8 | 5.0 | 11 | 5.7 | 86.9 |
| 2001 | 6.5 | 5.7 | 32 | 56.5 | 16.2 | 4.9 | 11 | 5.7 | 84.7 |
| 2002 | 5.2 | 4.3 | 30 | 54.2 | 16.9 | 5.1 | 11 | 5.3 | 95.2 |
| 2003 | 4.8 | 3.4 | 30 | 51.0 | 16.9 | 5.2 | 11 | 5.6 | 91.8 |

* The net amount of waste landfilled is the total (or gross) amount of waste landfilled that is corrected for the amount of construction waste landfilled. This construction waste is applied in a useful way, such as improvements of the infrastructure of the landfill or as a cover of the landfill (AOO, 2004a).

Source: AOO.

### 3.2.2 Policy measures in the waste market

In the last decade, a number of environmental policy measures have been undertaken that have affected the waste market. The policy measures listed below each have the potential to have a profound effect on the waste market and the supply and disposal of waste. In this subsection we will describe the potential effects of the policy measures.

[11] Nevertheless, at the time of writing a number of new incineration plants are under construction.

*Table 3.3 Environmental policy with respect to waste management*[12].

| | |
|---|---|
| 1994 | Harmonisation of waste trading in EU (implementation Waste Shipment Regulation, WSR) |
| 1994 | Separate collection of organic municipal solid waste |
| 1995 | Landfill ban: prohibition of landfilling recyclable waste and combustible waste (still with many exemptions) |
| 1996 | Implementation of the landfill tax on recyclable and combustible waste |
| 1998 | Establishment of a new after-care system for landfills under provincial responsibility), financed by a charge on landfilling at province level, see AOO (2000b) |
| 1999 | Implementation of an advanced disposal fee on new electric and electronic household equipment |
| 2000 | Provincial self-sufficiency for waste incineration abolished resulting in a national incineration market |
| 2001 | Source separation responsibility for producers (e.g. for waste paper) |

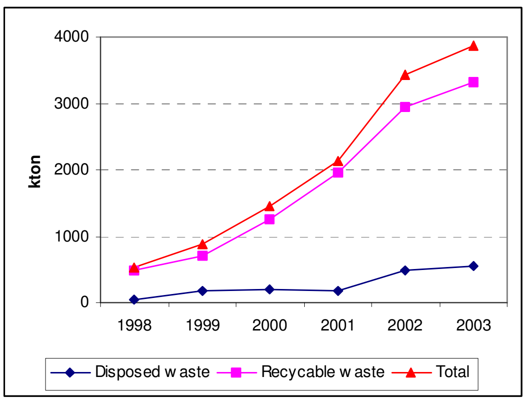


*Figure 3.3 Export of notified waste.*

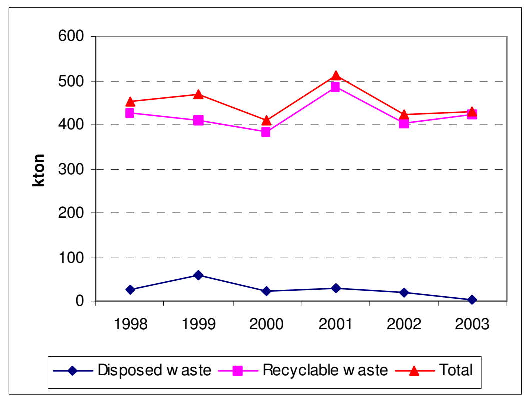


*Figure 3.4 Import of notified waste.*

*Harmonisation of waste trading in the EU*

In 1994, the European Waste Shipment Regulation (WSR, 259/93) came into force. With the WSR, the procedures of shipping waste between EU countries are harmonized. The WSR demands a notification of waste for disposal and hazardous waste at the competent authorities in the importing and exporting countries involved. Figure 3.3 and 3.4 show the development of exports and imports of notified waste[13] in the Netherlands for the period 1998-2003. The Netherlands is a net exporter of waste, and the export of waste is still growing. In 2003, approximately 85% of exported waste was recyclable waste. The trade in non-hazardous waste for recycling, which is more or less free, is of a much larger scale. Put into perspective: Import and export in The Netherlands of waste paper, an important waste stream in this category, amounts to approximately 2.5 million tonnes per

[12] Table 1.1 of Dijkgraaf (2004) summarizes several policy measures including some EU directives. In this study, however, we will focus only on the Dutch policy measures.

[13] The imports and exports of waste in these figures only refer to the waste products for which the report of transportation is compulsory. Transports of recyclable products, for instance, do not have to be reported.

year. A large part of the non-hazardous waste is is exported for recycling to South-East Asia.[14]

*Separate collection of organic and municipal solid waste*

Since 1994, municipalities have the obligation to collect organic waste and rest waste separately. This gave an enormous boost to the amount of household waste composted. Since the year 2000 several municipalities, especially the large ones like Utrecht and Rotterdam, have asked for and got an exemption from this law. They do not have to collect waste separately in some parts of the municipality if the quality of collected organic waste is too low.

*Landfill ban*

Since the 1st of October 1995 there exists a ban on the landfilling of recyclable and combustible waste.[15] This should have reduced the amount of landfilled waste substantially. However, the available capacity of alternative waste disposal options, incineration in particular, was insufficient in particular parts of the Netherlands (Limburg), so that the government had to allow municipalities and producers to landfill combustible waste despite the ban. In later years, when the capacity of the alternative options was increased it became less easy to get an exemption on the landfill ban.

*Landfill tax*

The landfill tax was introduced in 1995. Since 1998 there is a high rate for combustible waste, and a lower rate for non-combustible waste. Table 3.4 shows that the high rate started at a fairly low level and has been increased substantially over the last decade. In 2000 the high landfill tax rate was more than doubled, so that the price of incineration became lower than the price of landfilling.

*Table 3.4 Landfill tax in € per tonne.*

| Type of waste | 1995-1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 |
|---|---|---|---|---|---|---|---|---|---|
| Combustible waste | 13.25 | 29.13 | 29.75 | 64.28 | 65.44 | 78.81 | 81.65 | 83.61 | 84.78 |
| Non-combustible waste | 13.25 | 13.25 | 13.53 | 12.38 | 12.61 | 13.00 | 13.47 | 13.79 | 13.98 |

Source: Wet belastingen op milieugrondslag (Environmental taxes act), art. 18.

*Provincial responsibility of landfills*

On the 1st of April 1998 the provinces instead of the national government became responsible for preventing environmental leaks caused by landfills after the closure of landfills. With a view to this responsibility, the provinces introduced a charge on landfilling (see AOO, 2000b). The revenues of the charges are put into a fund and are used to prevent environmental damages after a landfill closes. Since the charge will probably

14 See FNOI website: www.fnoi.nl.

15 The definition of combustible waste is rather technical. By definition, all solid household waste is combustible.

raise the price of landfilling (landfill sites are free to include the charge in the landfill price) it can potentially lower the demand for landfilling.

*Advanced disposal fee on electr(on)ic household equipment*

Since 1999, consumers pay a certain advanced disposal fee ('verwijderingsbijdrage') when they purchase new electric and electronic equipment. Retailers are obliged to take back the 'disposed' product when the consumer purchases a new product. Consumers can also bring the product to a recycling centre or municipality free of charge. By formalizing the recycling process of electronic household equipment it becomes easier to recycle these products. Simultaneously with the implementation of the advanced disposal fee, a ban on landfilling and incineration of electronic household equipment came into force. In 2003, the advanced disposal fee amounted to €17 for freezers and refrigerators and €5 for washing machines.

*Provincial self-sufficiency for waste incineration abolished*

Since 2000, the regulation on the provincial cross-border transportation of recyclable and combustible waste has been relaxed. (see AOO, 2000b). Municipalities and producers have more freedom to choose an incineration plant or recycling firm regardless of the location. This could mean that the difference in waste disposal costs will become less between different provinces.

*Source separation responsibility for producers*

In 1999, the program separation responsibility for producers started. In 2001 the rules of this program were formalized in the National Waste Management Plan (LAP). Producers are obliged to separate all feasible waste materials. They should at least separate hazardous waste, paper and electronic appliances.

## 3.3 Determinants of waste supply and disposal options

### 3.3.1 Waste supply

In order to estimate the impact of waste disposal costs on total waste supply, we have to identify the main determinants of waste supply. Total waste supply is regressed on determinants, such as economic growth and population growth and the level of costs of waste disposal options. Also, we carry out a similar analysis for household waste and service sector waste. In the case of household waste, additional explanatory variables will be the household waste (disposal) charge rate and the growth of single person households. In the case of service sector waste, we include the relative costs of landfilling (including the landfill tax) over incineration, the number of firms in the service sector (instead of population growth), and the share of service waste collected by municipalities.

The regression model will look as follows: total waste supply is regressed on economic growth, population growth and the relative price of incineration and landfilling.

$$Y_{i,t} = \alpha_{i,t} + \beta^X X_{i,t} + \beta^W W_{i,t} + \beta^P P_{i,t} + \varepsilon_{i,t}, \tag{3.1}$$

Where $Y_{i,t}$ is total waste supply at time t for sector/region $i$, $X_{i,t}$ includes demographic and economic determinants, such as Gross Domestic Product (GDP), population growth and other, $W_{i,t}$ includes indicators of other policy measures such as the (rest) capacity of waste disposal options, and $P_{i,t}$ is the (relative) price of waste disposal option at time t for sector/region *i.* The coefficients to be estimated are $\alpha$, $\beta^X$, $\beta^W$ and $\beta^P$, while $\varepsilon_{i,t}$ is the residual. Specifically, the coefficient $\beta^P$ indicates what the impact of (relative) prices of disposal options is on the total waste supply in the Netherlands. If $\beta^P$ refers to the price of landfilling, and $\beta^P$ <0 and significant, then a higher landfill price (due to the landfill tax for instance) decreases the total waste supply.

This analysis will be applied on the household and service sector. Before we will analyse the determinants of waste supply for both sectors, we briefly summarize the developments of waste supply and the economic and demographic developments. In the case of the service sector we replace population growth by the growth in the number of firms for instance.

In the case of the households, we have household data at province level, which seems an appropriate division taking into account the ban on provincial cross-border waste transportations in the last decade. In the case of households, we will replace waste disposal costs by the waste disposal charge, since households are confronted with this charge and not with landfill taxes directly. Especially, only if a municipality levies a unit-based waste disposal charge, the landfill tax can be of direct influence on the supply of household waste.

For the service sector, we cannot make a geographical division of waste, because the information on the supply of service waste is unavailable at province level. In the analysis of the waste of the service sector, the (relative) prices of disposal options (landfill costs versus incineration costs) are important, because a substantial part of the firms have contracts with waste disposal companies. Obviously, we will include the number of firms instead of population.

### 3.3.2 Waste disposal options

From Section 3.3.1, we get insight into the determinants of total waste supply (per sector), and the impact of waste disposal prices (including landfill taxes) on the total waste supply (per sector). However, we now take a closer look at the impact of the landfill tax on the choice between recycling and the waste disposal options.

Most ideally, we would like to analyse this impact at the level of agents (specialized collection companies such as ROVA and SITA, or municipalities) that are responsible for choosing recycling or waste disposal options. However, information is not available at such a detailed level. Therefore, we use similar kind of data at the same level as in the waste supply analysis described in Section 3.3.1.

This analysis is divided into two phases. Firstly, we start with the analysis of the ‘choice’ between recycling and waste disposal (the total of incineration and landfilling). The share of recycling is regressed on a number of determinants such as the price level of waste disposal options. In this way, we identify the impact of waste disposal prices on the share of recycling in total waste supply.

$$S_{i,t} = \alpha_{i,t} + \gamma^{X} X_{i,t} + \gamma^{W} W_{i,t} + \gamma^{P} P_{i,t} + \varepsilon_{i,t}, \tag{3.2}$$

where $S_{i,t}$ is the share of recycling in the total waste supply at time t for sector/region *i*, $X_{i,t}$ is a set of geographical/sectoral information and economic determinants, and $W_{i,t}$ is a set of indicators for policy measures such as the rest capacity of waste disposal options and the law on separated collection of waste, import-export of recyclable waste and the implementation of the disposal surcharge on electr(on)ic household appliances. In addition, $P_{i,t}$ is a set of financial indicators such as the costs of disposal options, including landfill taxes. In the case that the costs of landfilling are included, a positive coefficient of this variable means that *ceteris paribus* the share of recycling increases if the landfill tax is increased. This analysis is explored for both sectors (households and service sector), and the set of explanatory variables differs between both sectors.

Secondly, we model the choice between incineration and landfilling in order to analyse the impact of the landfill tax (and landfilling costs) on the waste disposal choice.

$$D_{i,t} = \alpha_{i,t} + \delta^{X} X_{i,t} + \delta^{W} W_{i,t} + \delta^{P} P_{i,t} + \varepsilon_{i,t}, \tag{3.3}$$

where $D_{i,t}$ is the total amount of waste supplied to a specific disposal option (incineration or landfilling) at time *t* for sector/region *i*. The set $X_{i,t}$ includes relevant geographical/sectoral information, and the set $W_{i,t}$ consists of indicators for policy measures such as the rest capacity of waste disposal options, the landfill ban and the opening of province borders for waste transport (in fact, the regulations of cross-border transportations were relaxed). In the case of the latter two, there are no quantity indicators for these policy measures. We therefore use dummy variable indicators for periods in which particular policy measures were present. Note however that we do not have the intention to estimate the impacts of other policy measures, but we try to obtain unbiased estimates for the characteristics we can quantify in the analyses.

In addition, $P_{i,t}$ is a set of financial indicators such as (relative) price of recycling and different waste disposal options, including landfill taxes, provincial taxes, and transportation. If the costs of landfilling are included and the coefficient has a negative sign, this means that a higher landfill tax (which increases the price of landfilling) reduces the share of landfilling. This analysis is explored for both sectors (households and service sector), and the set of explanatory variables differs between both sectors.

## 3.4 Data and results

For the ex-post assessment, we composed two separate data sets for household waste and waste from the service sector. The household data are collected at province level, and the service sector data at sub-sector level. Variables reflecting monetary values are expressed in the price level of 2000. In the case of disposal costs, i.e. the costs of landfilling and the costs of incineration, we ignore transportation costs, because there are no reliable data on transportation costs. Dijkgraaf *et al.* (2001) summarize transportation costs for incineration plants, but these estimates are highly aggregated. Also AOO (2003e and 2004a) present incineration costs including and excluding transportation costs although not systematically. We tried to derive a time series on transportation costs, but this time series was inconsistent. Moreover, we do not have transportation costs for landfilling. Furthermore, the transportation costs reported in AOO (2003e, 2004a) indicate that the transportation costs account for a small proportion of the total disposal costs. Since there are more landfills present than incineration plants, we presume that the transportation costs of landfills will not exceed the transportation costs of incineration.

Both data sets consist of sectors/regions with time series of observations. Therefore, we apply panel data regression techniques, such as Fixed Effects (FE) estimation or Random Effects (RE) estimation. We choose these techniques, because these estimation techniques are straightforward, and both estimation methods take into account heterogeneous (un)observed effects. As a consequence, we can obtain unbiased estimates for the coefficients of the variables that are included in the analyses. Determinants that are constant over time for regions (surface of regions for instance) or sectors (location of firms, for instance) cannot be taken into account. FE estimation corrects for region or sector specific information over time while RE also takes into account differences in information across regions or sectors. Altough the RE estimation seems more complete, the choice of the panel data estimation technique is not straightforward, because each analysis has different requirements. Hsiao (1986) argues that the choice between FE and RE has to be based on intuition. Intuitively, FE estimation is more suitable if there are autonomous regional or sectoral differences in the data. RE estimation is suitable if there is no reason to assume those differences.

In the case of the household waste analysis, it is likely that there are regional differences due to differences in available disposal capacity and strict regulation of waste transports, for instance. For the service sector, there might be differences in recycling (glass in the hotel and catering industry), for instance, but these sectoral differences are less clear. RE estimation will suffice. Moreover, due to lack of data FE turned out to be infeasible.

In order to to correct for trends in population and number of firms, we analyse per capita and per firm waste indicators for the household data analysis and the service sector analysis respectively.

### 3.4.1 Household waste

The household data are collected for 12 provinces over the period 1995-2003. Thus the dataset contains 108 records. Table 3. 5 shows the variables in the dataset for household waste. The dataset consists of information on the number of sites/plants, (rest) capacity, the amount of waste disposed and tariffs of the disposal options at province level. If necessary, indicators at higher aggregation level can be constructed. For instance, some provinces do not have waste incineration plants within their borders and despite the strict regulation on transportation of waste over the province borders, provinces with insufficient incineration capacity can get exemptions to transport combustible waste to incineration plants or to landfill combustible waste.

The socio-economic variables include volume deflators of Gross Domestic Product (GDP), population and the number of (single person) households, all at province level. Although we would have preferred final private consumption (FPC) as an economic indicator instead of GDP, the indicator FPC is not available at the level of provinces. Additionally, we have the number of municipalities with and without any form of unit-based pricing including the average waste disposal charge. However, we only have this information for the period 1998-2003. For the earlier period, we use national averages of waste disposal charges.[16] With the use of year dummies we can correct the regression results for this omission. For the waste disposal charge we know the average at national level for the period prior to 1998.

Table 3.6 presents the estimation results of household waste generation and disposal options. All equations (3.1), (3.2) and (3.3) are estimated with a Fixed Effects (FE) regression, which takes into account the possible (unobservable) heterogeneity between provinces. For convenience, we ignore the coefficients of the fixed effects themselves, and focus on the other determinants.

The first column of Table 3.6 presents the determinants of the generation of waste per capita as in eq. (3.1). In this regression we included waste disposal charge instead of actual disposal costs, because the generation of household waste does not directly depend on the costs of disposal options. Merely, the generation of the household waste is more likely to depend on the level of the waste disposal charge. However, if the waste disposal charge follows a flat fee scheme (not depending on a characteristic of supply, such as frequency, volume, or weight), then it is unlikely that the waste disposal charge has any effect on the generation of household waste. In the case of a unit-based pricing scheme, the disposal costs can affect the unit-based price, and consequently this might affect the generation of waste. On average, the disposal charges in municipalities with a unit-based pricing scheme are lower than in municipalities with flat fee pricing schemes (AOO, 2003e). Moreover, Dijkgraaf and Vollebergh (2004a) argue that the amount of waste generated is less as well for different kinds of unit-based pricing schemes. They estimated the impacts of different kinds of regimes of waste disposal charging with a regression model using data of Dutch municipalities. For this reason, we include the share of municipalities with a unit-based pricing scheme into the analysis.

The determinants of household waste generation per capita are analysed with Eq. (3.1). As mentioned above, the generation of household waste is not directly affected by costs of disposal options, and the impact of the waste disposal charge is likely to occur if a province has a large share of municipalities with unit-based pricing schemes. The results in the first column of Table 3.6 are in line with our expectations. The waste disposal charge has no effect on waste generation, while provinces with higher shares of municipalities with unit-based pricing regimes tend to generate less waste per capita. Furthermore, higher levels of GDP per capita lead to higher total amounts of household waste per capita. Finally, the number of singles has a positive effect on household waste per capita.

To explain the share of recycling as in Eq. (3.2), we include the level of disposal costs explicitly, because we are analysing the disposal option recycling versus incineration and landfilling. The disposal costs (costs of incineration and landfilling including landfill tax) are simply defined as the unweighted average of those two costs. Furthermore, we constructed disposal capacity, which is the total capacity of incineration and landfilling. Since landfilling capacity is measured in volume, we use the weight per volume of 1,250 kg per cubic metre.

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16 Before 1998, there was only a small number of municipalities that had introduced unit-based pricing regimes.

The second column of Table 3.6 shows that the share of recycling in household waste increases with the level of per capita GDP, and there are some autonomous changes throughout the period. The share of recycling was higher in the 1995 and in the period 1996-1997 if we correct for the other determinants. Note that this does not mean that the total amount of recycled waste declined, which is only the case if the total level of household waste increased at a lower pace than the reduction in the recycling share.

The third and fourth columns show the explanation of the amount of landfilled and incinerated household waste respectively according to the regression in Eq. (3.3). In these analyses, we include the relative price of landfilling over incineration.

GDP per capita has a positive effect on the total amount of landfilled household waste, and in 1995 the amount of landfilled household waste was significantly higher. Provinces with a higher share of municipalities with a unit-based pricing regime have a significant lower level of waste landfilling. The relative price of landfill costs has no significant effect.

From the incineration analysis, we observe a significant positive effect of waste incineration by provinces with a higher share of municipalities with a unit-based pricing regime. Moreover, the availability of incineration capacity encourages waste incineration as well. This indicates that if there is sufficient incineration capacity available, municipalities will incinerate household waste instead of dumping it in landfills. Municipalities have a preference for incineration over landfilling because of the ban on landfilling household waste and the existence of long-term supply contracts between municipalities and waste incineration plants. In addition, municipalities often hold shares in waste incineration plants, see AOO (2000a). Furthermore, the relative costs of landfilling have no significant effect on the amount of household waste landfilled or incinerated. The waste disposal choice of municipalities or their contracted collectors is not affected by the higher landfill costs due to the high landfill tax.

The results of Table 3.6 indicate that waste disposal charges did not affect the generation of household waste, and the costs of disposal options did not affect the amount of waste recycled, landfilled or incinerated. Basically, there is no effect, because changes in disposal costs are not internalised in the waste disposal charges as displayed in Figure 3.5.

Figure 3.5 shows the developments in the real costs of landfilling (including landfill tax) and incineration in comparison with the average waste disposal charge paid per tonne of waste per household. The costs of landfilling show a steeply increasing trend caused by the development in the landfill tax. The incineration costs show a gradually increasing trend. Remarkably, the waste disposal charge is fairly constant over the period 1995-2003, despite simultaneous growths in the household waste per capita (see Figure 3.1), and in the disposal costs. This implies that the disposal costs are not fully internalised in the waste disposal charges. This is confirmed by the findings of AOO (2004a) that there is no statistical relationship between disposal costs and waste disposal charges. Note however that waste disposal charges are used for financing municipal waste management costs which also include costs of waste collection. Moreover, AOO (2003e) argues that 95% of the municipal waste management costs is covered by the revenues of the waste disposal charges. This ratio is fairly constant in the period 1995-2003.

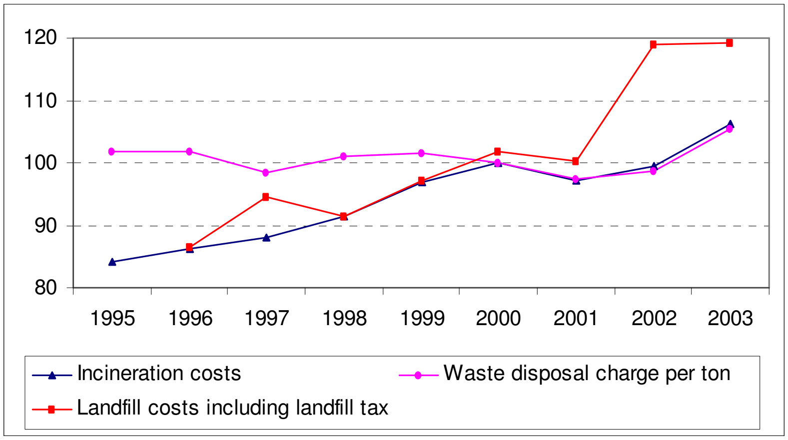


*Figure 3.5* *Development of incineration costs, landfilling costs (including landfill tax) and waste disposal charge in real terms in euros (2000) per tonne.*

### 3.4.2 Service sector

The dataset for the waste supplied by the service sector is smaller than the household dataset. In contrast with the household waste data, we distinguish 5 sub-sectors[17] within the service sector instead of spatial division, namely:

- Retail and wholesale market (including repair industry);
- Hotel and catering industry;
- Transportation and communication;
- Financial services and insurance industry; and
- Other services (public management, health care, education and cultural services).

Table 3.7 presents the summary statistics of the database on the service sector waste. The data cover a period of 8 years from 1995 to 2002. The amounts of waste per disposal option of the sub-sectors are derived from information on a broader definition of the service sector, which partly includes the manufacturing industry. As a consequence, the ratio between incineration and landfilling waste is similar for all sub-sectors. Therefore, we analyse the amounts of waste landfilled and incinerated instead of the share of landfilling and incineration.

Table 3.8 presents the estimation results of the service sector. The waste generation is estimated with Ordinary Least Squares (OLS), while for the other indicators we use the Random Effects (RE) panel data estimation. Sector-specific features are now included as a stochastic term instead of a dummy variable as in the Fixed Effects estimation. Preferably, all equations were estimated with Fixed Effects estimation to correct the estimation results for unobserved heterogeneous effects. However, in the case of the waste

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[17] Instead of the five sub-sectors distinguished in this study, AOO (2004b) distinguishes 7 sub-sectors (viz. separate sectors for wholesale market, retail market sector and repair industry). Due to the lack of economic data for these 7 sub-sectors, we had to merge these sub-sectors. Originally, AOO (2004b) used the SBI'74 classification for the sub-division of the service while we prefer the more up-to-date SBI'93 classification. Fortunately, the SBI'74 and SBI'93 classifications of the sub-sectors of the service sector do not differ much.

supply regression estimation (3.1), both FE and RE estimation were rejected. In the other cases, Equations (3.2) and (3.3), FE estimation was rejected while RE estimation was not. The main reason for rejecting FE estimation is the limited numer of observations in this analysis ($N$=40). Again, we are not specifically interested in sector-specific effects but we are interested in unbiased estimates of the other determinants.

The waste generation equation, as in Eq. (3.1), shows that except for the sector dummies the employment per firm is the only significant determinant. Higher levels of employment imply higher levels of waste generation per firm. In the estimation of the recycling share of the waste sector according to Eq. (3.2), the level of the disposal costs (average of incineration costs and landfilling costs) has a significant positive impact on the share of recycling. Higher levels of disposal costs imply a higher share of recycling in the service sector.

From the estimations of the amount of landfilled and incinerated waste as in Eq. (3.3), we observe that the relative price of landfill costs over incineration costs has a negative effect on the amount of waste landfilled, and a positive effect on the amount waste incinerated. Higher levels of landfill tax will increase the relative price of landfill costs over incineration prices, and as a consequence, the service sector will use the incineration option more and the landfill option less.

## 3.5 Conclusions

This chapter discusses the possible impacts of the landfill tax on household waste and service sector waste. Within an ex-post assessment, four research questions were addressed. First of all, has landfilling become a more expensive alternative of waste disposal than incineration and recycling? Although the landfill tax was introduced in 1995, landfilling became more expensive than incineration only around the year 2000 as a result of the increased tax rate. Whether or not landfilling is more expensive than recycling is less clear, because there are various types of recycling (such as glass, paper, textiles, etc). Moreover, the choice between recycling and disposing waste is taken at a different level, i.e. the waste generator and not the waste disposer. This is especially true in the case of household waste. Households choose between recycling or disposing waste, while municipalities choose between incinerating and landfilling. For the service sector, individual companies take both decisions.

The second question was if the landfill tax has led to larger amounts of waste to be incinerated and recycled, and to less waste to be landfilled. With the regression techniques, this question was handled. The results of the analysis of household data are uniform. The landfill tax did not have a significant direct impact on the generation of household waste, nor did it affect the choice for waste disposal options. In the case of household waste generation, the increases in the waste disposal costs, such as the landfill tax, are not internalised in the waste disposal charges in the case of flat fee regimes. In the case of unit-based pricing regimes, the disposal costs are more likely to be internalised in the waste disposal charges. However, to analyse this indirect effect requires an additional analysis of the impact of waste disposal costs on the tariffs of unit-based pricing regimes at the level of municipalities. The presence of a unit-based pricing regime decreases household waste generation. In addition, it decreases the amount of waste landfilled, while it increases the amount of waste incinerated. This latter result is somewhat surprising, as it is not the households themselves who decide on the choice between landfilling and incineration, but rather the municipalities (or their contracted collectors of household waste). Moreover, in the case of household waste the functioning of the market mechanism is restricted by the landfill ban and existing long-term contracts on waste supply with waste incineration plants.

For the service sector, the level of disposal costs is not affecting the generation of waste, although it affects the waste disposal choice. In particular, higher costs for landfilling and incineration increase the share of recycling. Moreover, if the relative increase of costs of landfilling exceeds the relative increase of costs of incineration, firms from the service sector will landfill less waste and incinerate more. In this sense, the landfill tax can play a crucial role in the decision making of firms from the service sector with respect to disposing waste

The third question dealt with the efficiency of use of the waste incineration plants and recycling infrastructure. On the one hand we observe that the capacity of incineration has been fairly constant since 1997. The number of waste incineration plants was constant, and only little fluctuations in capacity were observed. On the other hand, the total amount of waste incinerated increased slightly in the corresponding period, so that the efficiency (capacity utilisation) of waste incineration plants shows a positive trend from 80% in 1995/1996 to more than 90% in 2002/2003 (see Table 3.2). In addition, the results of the regression analysis showed that the relative price of landfilling tariffs and incineration costs had a positive effect on the amount of service sector waste that has been incinerated. This means that higher costs of landfilling due to higher landfill tax levels will increase the demand of the service sector for the incineration of waste. Due to a constant capacity and a slightly growing incineration of waste, the efficiency of use of waste incineration plants shows a slightly positive trend. In the case of recycling, we observed a moderate growth in the case of household waste, and a strong growth of recycling in the case of the service sector waste. We were not able to link these findings to the efficiency of the use of recycling infrastructure.

Finally, until 2003 the landfill tax has not led to new investments in new incineration plants in The Netherlands. However, export of waste for incineration and use as a secondary fuel has increased (see Section 3.2.1). So, the landfill tax in the Netherlands might have contributed to a more efficient use of foreign waste incineration capacity and/or new foreign incineration capacity. More research into developments of the European waste market is needed to come to a reliable conclusion on this issue. If the landfill tax has led to investments in recycling infrastructure is ambiguous, because different kinds of recycling options require their own infrastructure. In future research, the development of recycling options could be mapped in more detail. The large and increasing export of waste paper to South-East Asia (cf. Van Beukering, 2001) suggests that global transport and recycling industries benefit from the source separation efforts of consumers and companies in The Netherlands (and Europe as a whole).

The evaluation of impacts of the landfill tax has resulted in different results for different sectors, because many aspects play a role. If the disposal costs are reflected in the costs charged to the waste suppliers, as is the case in the service sector, the landfill tax turns out to be an effective instrument. Higher levels of the landfill tax will imply lower levels of waste landfilled and higher levels of waste incinerated or recycled. If changes in disposal costs are not incorporated in the charges to the waste generators (as is the case of households), the landfill tax seems to be ineffective, especially if there is a landfill ban imposed as well. In that case, higher landfill taxes are not or delayed internalised in waste disposal charges. The introduction of unit-based pricing for household waste collection might provide the correct incentive in which higher landfill taxes are internalised.

*Table 3. 5 Summary statistics of dataset on household waste, 1995-2003.*

| Variable description | N | Min | Max | Mean | Std. Dev |
|---|---|---|---|---|---|
| Remaining landfill capacity per province (in million | 108 | 0.3 | 17.7 | 5.4 | 3.9 |
| Landfill tariffs including landfill tax (euros per ton) | 108 | 54.4 | 119.2 | 97.3 | 18.0 |
| Landfill tax (euros per ton) | 108 | 0 | 74.3 | 44.3 | 24.0 |
| Waste incinerated per province (kton) | 108 | 0 | 1798 | 372.8 | 534.8 |
| Incineration capacity per province (kton) | 108 | 0 | 1875 | 353.9 | 546.5 |
| Rest capacity of incineration plants (kton) | 108 | 254 | 1075 | 658.0 | 228.1 |
| Incineration tariff (euros per ton) | 108 | 91.8 | 115.6 | 102.8 | 7.5 |
| Household waste per province (kton) | 108 | 126 | 1810 | 706.1 | 485.3 |
| Disposed household waste (kton) | | | | | |
| Landfilled | 108 | 0 | 677.6 | 80.4 | 127.1 |
| Incinerated | 108 | 0 | 1105.2 | 241.6 | 294.3 |
| Recycled or composted | 108 | 27.2 | 947.9 | 406.7 | 239.9 |
| Population (x 1000) | 108 | 272.8 | 3451.9 | 1323.0 | 961.7 |
| Population density | 108 | 172 | 1225 | 475.0 | 323.7 |
| Total number of households (x 1000) | 108 | 103.8 | 1541.7 | 566.2 | 426.5 |
| Number of single person households (x 1000) | 108 | 25.2 | 574.5 | 189.5 | 160.9 |
| Gross Domestic Product (GDP) in mln euro | 108 | 3951 | 100270.4 | 31273.7 | 25440.1 |
| Annual volume growth of GDP in % | 96 | -2 | 9.4 | 2.5 | 2.4 |
| Annual deflation of GDP in % | 96 | -2 | 11.3 | 2.6 | 1.9 |
| Number of municipalities per province | 72 | 6 | 95 | 43.0 | 27.1 |
| Without unit-based schemes | 72 | 5 | 92 | 33.2 | 23.8 |
| With unit-based schemes | 72 | 0 | 36 | 9.8 | 11.3 |
| Average waste disposal charge in euro (2000) per household | 108 | 158.3 | 228.1 | 192.3 | 14.6 |

Sources: CBS, AOO, RIVM.

*Table 3.6 Results of the regressions of household waste per capita, share of recycled household waste, landfilled household waste per capita, and incinerated household waste per capita (standard errors in parentheses).*

| | Eq. (1) | | Eq. (2) | | Eq. (3) | | Eq. (3) | |
|---|---|---|---|---|---|---|---|---|
| | FE<br>Waste supply | | FE<br>Recycled waste (share) | | FE<br>Landfilled waste | | FE<br>Incinerated waste | |
| Intercept | 2.974 | | 1.507 | # | 0.429 | | 0.443 | |
| | (6.058) | | (0.854) | | (0.583) | | (0.495) | |
| GDP per capita | 0.014 | ** | 0.015 | | 0.018 | * | -0.003 | |
| | (0.003) | | (0.01) | | (0.007) | | (0.007) | |
| Share of mun. with unit-based pricing scheme | -0.087 | ** | -0.008 | | -0.148 | * | 0.164 | ** |
| | (0.027) | | (0.086) | | (0.061) | | (0.058) | |
| Wast disposal charge (x € 1000) | 0.168 | | | | | | | |
| | (0.257) | | | | | | | |
| Price level of disposal costs in € | | | -2.393 | | | | | |
| | | | (1.828) | | | | | |
| Relative price of lanfdilling over incineration | | | | | -0.045 | | 0.018 | |
| | | | | | (0.108) | | (0.105) | |
| Share of single persons | 1.774 | * | -4.007 | | -2.352 | | -0.936 | |
| | (0.894) | | (2.808) | | (1.859) | | (1.605) | |
| Landfill capacity (x 1000) | | | | | 1.701 | | | |
| | | | | | (1.141) | | | |
| Incineration capacity | | | | | | | | |
| Incineration rest capacity (x 1000) | | | | | | | 0.015 | ** |
| | | | | | | | (0.004) | |
| Total disposal capacity | | | 0.001 | | | | | |
| | | | (0.001) | | | | | |
| Trend | -0.002 | | | | | | | |
| | (0.003) | | | | | | | |
| 1995 | | | 0.144 | ** | 0.113 | ** | -0.027 | |
| | | | (0.039) | | (0.038) | | (0.037) | |
| Period 1996-1997 | | | 0.125 | ** | 0.019 | | 0.051 | * |
| | | | (0.03) | | (0.204) | | (0.019) | |
| Period 1995-1998 | -0.029 | ** | | | | | | |
| | (0.01) | | | | | | | |
| $R^2$ within | 0.69 | | 0.69 | | 0.64 | | 0.37 | |
| *N* | 12 | | 12 | | 12 | | 12 | |
| *T* | 9 | | 9 | | 9 | | 9 | |
| *K* | 20 | | 20 | | 20 | | 20 | |
| F( *k*-1,*N*-*k*) = | 33.67 | ** | 28.14 | ** | 22.26 | ** | 7.49 | ** |
| F-test on fixed effects F(*N*-1,*N*-*k*) = | 28.56 | ** | 50.18 | ** | 15.17 | ** | 27.27 | ** |

** at 1% significance level, * at 5% significance level, and # at 10% significance level.

*Table 3.7 Summary statistics of dataset on waste supplied by the service sector, 1995-2002.*

| Variable description | N | Min | Max | Mean | Std. Dev |
|---|---|---|---|---|---|
| Total waste (in kton)) | 40 | 186 | 1668 | 696.4 | 461.9 |
| Waste collected separately (in kton) | 40 | 67 | 810 | 293.3 | 227.5 |
| Waste not collected separately (in kton) | 40 | 93 | 891 | 403.1 | 243.4 |
| Waste for recycling (in kton) | 40 | 84.3 | 740.8 | 293.4 | 200.4 |
| Waste incinerated (in kton) | 40 | 37.3 | 538.8 | 159.8 | 113.4 |
| Waste landfilled (in kton) | 40 | 61.3 | 602.9 | 243.2 | 165.6 |
| Total amount of waste landfilled in the NL in Mton | 40 | 5.2 | 9.9 | 7.3 | 1.3 |
| Landfill capacity in mln cubic meters | 40 | 54.2 | 79.3 | 66.2 | 9.3 |
| Number of landfills in NL | 40 | 30 | 47 | 39.3 | 6.0 |
| Landfill tariff including landfill tax in euros (2000) | 40 | 48.9 | 127.9 | 93.0 | 21.3 |
| Landfill tax in euros (2000) per ton | 40 | 0 | 78.8 | 40.8 | 24.8 |
| Incinerated waste in kton | 40 | 2811 | 5088 | 4385.4 | 765.2 |
| Incineration capacity in kton | 40 | 3350 | 5732 | 5067.8 | 742.1 |
| Rest capacity of incineration plants | 40 | 254 | 1075 | 682.4 | 232.4 |
| Number of incineration plants in NL | 40 | 7 | 11 | 10.3 | 1.4 |
| Incineration tariff in euros (2000) per ton | 40 | 91.8 | 108.9 | 101.2 | 6.4 |
| Value added (VA) in mln euro | 40 | 5254 | 108545 | 48281.6 | 32369.3 |
| Annual volume growth of VA in % | 35 | -3.7 | 10.4 | 3.7 | 3.2 |
| Annual deflation of VA in % | 35 | -4.8 | 10.3 | 2.5 | 3.0 |
| Employment in thousand men years | 40 | 130.7 | 1672.0 | 792.9 | 507.5 |
| Number of firms | 40 | 27,595 | 205,545 | 105,904.5 | 63,618.3 |

Sources: AOO, CBS, RIVM.

*Table 3.8 Results on the regressions of service sector waste per firm, share of recycled waste from the service sector, landfilled waste per firm, and incinerated household waste per firm.*

| | Eq. (1) | Eq. (2) | Eq. (3) | Eq. (3) |
|---|---|---|---|---|
| | OLS | RE | RE | RE |
| | Waste supply | Recycled waste (share) | Landfilled waste | Incinerated waste |
| Intercept | 0.003 | 0.088 | 0.006 ** | 0.002 * |
| | (0.002) | (0.061) | (0) | (0) |
| Value added per firm (x 1000) | | -0.072 | -0.026 * | -0.016 * |
| | | (0.229) | (0.01) | (0.007) |
| Employment per firm | 0.931 * | 0.414 | -0.032 | -0.020 |
| | (0.355) | (1.308) | (0.038) | (0.029) |
| Price level of disposal costs | 0.004 | 3.373 ** | | |
| | (0.015) | (0.612) | | |
| Relative price of landfilling over incineration (x1000) | | | | 1.045 * |
| | | | | (0.48) |
| Disposal capacity | | -0.003 | | |
| | | (0.007) | | |
| Incineration capacity | | | | 0.003 |
| | | | | (0.002) |
| Landfill capacity | | | 0.454 * | |
| | | | (0.195) | |
| 1995 | | -0.029 ** | 0.250 | -0.034 |
| | | (0.01) | (0.183) | (0.182) |
| Period 2000-2002 | | 0.006 | 0.136 | 0.166 |
| | | (0.0101) | (0.154) | (0.151) |
| Sector 130 | 0.002 ** | | | |
| | (0) | | | |
| Sector 150 | -0.006 * | | | |
| | (0.002) | | | |
| Sector 160 | -0.006 ** | | | |
| | (0.001) | | | |
| Sector 170 | -0.010 ** | | | |
| | (0.003) | | | |
| Adjusted $R^2$ | 0.84 | | | |
| $R^2$ within | | 0.74 | 0.64 | 0.28 |
| *N* | 5 | 5 | 5 | 5 |
| *T* | 8 | 8 | 8 | 8 |
| *K* | 7 | 7 | 7 | 7 |
| F( $k$-1,$N$-$k$) = | 34.9 | 0 | 0 | 0 |
| F-test on fixed effects F($N$-1,$N$-$k$) = | | | | |
| Wald Chi$^2$ ($k$-1) | | 94.1 ** | 72.48 ** | 23.19 ** |

** at 1% significance level, * at 5% significance level,and # at 10% significance level.

# 4. Ex-ante evaluation of the landfill tax

## 4.1 Introduction

The *ex-ante* analysis presented in this chapter addresses the following research questions (introduced as questions 1e, 1f, 3a and 3b in Chapter 1):

- What are the conditions that the waste market has to fulfil in order to apply the landfill tax as an effective instrument?
- What is the desired or optimal rate of the waste tax in order to make the suppliers of waste choose alternative waste treatment options (incineration and recycling)? In particular, should the rate be differentiated according to certain aspects such as type of waste, treatment option and waste supplier?
- What are the financial consequences (in the short and the long term) of landfill taxation, landfill bans and legal obligations (for landfills and incineration plants) to accept waste, both for the waste suppliers and the waste treatment companies?
- Which instrument will (in the short and the long term) lead to the lowest costs of waste treatment for the waste suppliers?

In order to answer these questions, a number of scenarios are specified. These scenarios differ in terms of landfill tax rate and in terms of other waste market features, such as international trade in waste. The implications of each scenario are analyzed using a general equilibrium model. The model used is based on the model presented in Chapter 4 of Bartelings (2003). To evaluate the effectiveness of the landfill tax, production of waste by producers and international trade has been added to this model.

Section 4.2 describes the model. In Section 4.3, the dataset, the scenarios and the results of the model simulations are presented. In Section 4.4, the same model is used to compare the impact of a landfill tax to the impact of a ban on landfilling. Section 4.5 contains conclusions.

## 4.2 Description of the model

### 4.2.1 Introduction

The model presented here is a general equilibrium model in the Negishi format. General equilibrium models are economy-wide models in the sense that they cover all major economic transactions. The reason for modeling all relevant markets simultaneously is the existence of complex interactions in an economy. Partial models are based on the *ceteris paribus* condition, *i.e.* the remainder of the economy is assumed to be constant during policy simulations. As long as the ceteris paribus condition holds, partial models are fine, and the complications and data-requirements of general equilibrium models can be safely avoided. If, however, there are significant linkages between different markets, a partial analysis may lead to inaccurate and perhaps biased results due to the existence of indirect effects[18]. In an extreme case, the indirect effects, as captured by general equilibrium models, may outweigh the direct effects, as captured by partial models. This can result in opposite policy recommendations (Thissen, 1998).

General equilibrium models can be built in different formats, such as the Computable General Equilibrium (CGE) format, the Negishi format, the full format, and the open economy format. Each of these formats has its strengths and weaknesses (for more information see Ginsburgh and Keyzer, 1997). The model presented in this report is written in the Negishi format, which is especially suitable for the implementation of externalities, such as environmental pollution and waste generation; and price rigidities, like a zero marginal price for waste collection. Note that a format is just a way to build a general equilibrium model. The choice of a format does not influence the optimal solution, only the way of finding it.

In this Chapter we will first present a general overview of the model. Then we will explicitly go into the model techniques concerning international trade in waste and flat fee pricing of waste collection. Finally we will present the complete model structure.

### 4.2.2 General description of the model

In a simplified economy, two types of actors are distinguished: households and firms. Households consume goods and supply capital and labour; firms produce goods with the use of capital, labour and intermediate goods. Consumers are differentiated into two types: private consumers and the government. Five different production sectors are distinguished, together producing thirteen unique goods. These sectors are: (1) an extraction sector producing virgin material; (2) a production sector producing eight types of services and goods: 'Wholesale market and auctions', 'Retail market', 'Repairment industry', 'Hotel and Catering industry', 'transport services', 'Financial services', 'other services' and a sector producing all other goods in the economy called: 'rest economy'; (3) a recycling sector producing recycling services; (4) a collection sector producing waste collection services and (5) a waste treatment sector producing incineration services and landfilling services. The hypothetical economy is shown in Figure 4.1.

All firms use capital and labour, which are bought from the consumers, to produce material goods or services. The production sector of consumption goods also uses virgin material or recycled material. Consumption and production leads to waste generation. Producers can recycle waste by investing more capital and labour in the production process. Waste that is generated has to be transported to an incineration plant or landfilling site.

Consumption of services by private households leads to the generation of municipal solid waste. Waste must be either recycled or collected by the municipality. We differentiate two types of municipalities in this model. One municipality charges a flat fee for waste collection; the other municipality charges a unit-based price for waste collection. Comparing the results for the two different types of municipalities will show how the effectiveness of a landfill tax is influenced by the pricing mechanism for waste collection.

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[18] The indirect effects capture the interactions between different markets. Any change in one market can result in a change within another, which in turn can affect a change in the original market.

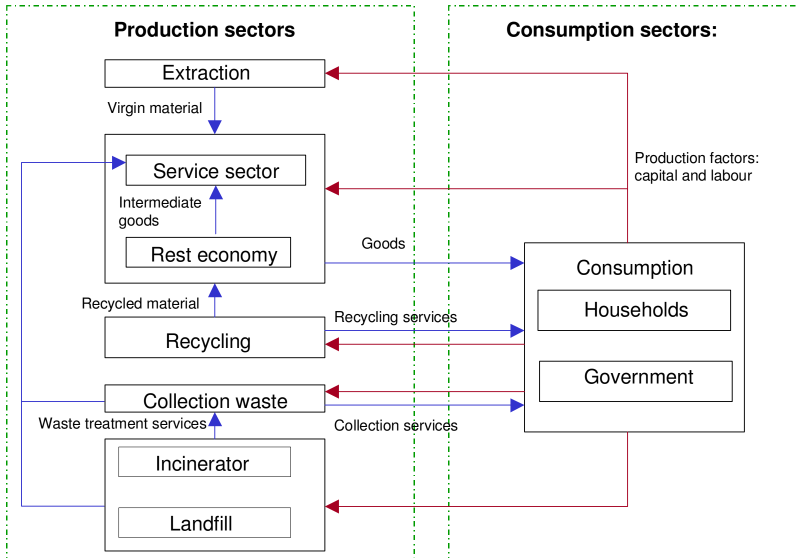


*Figure 4.1 Representation of the economy.*

We assume that collected rest waste is not separated and recycled after collection, but is instead sent immediately to an incineration plant or landfill unit. We are primarily interested in the choice the consumer makes: the consumer can, for example, choose to separate organic waste, paper, or glass from rest waste. The consumers will have to incur costs in order to separate or recycle these materials. Recycling will, for example, cost the consumer both time and storage space. This is modeled as if the consumer buys 'recycling services'. By buying recycling services, they generate recyclable waste; this waste is sent to a recycling unit where it is turned into recycled material. In the rest of this chapter we will use the term recycling for various activities the consumer can undertake to prevent rest waste: this includes waste separation in rest waste, glass, paper, and organic waste and composting.

Consumers can prevent waste by recycling more or, to a lesser extent, by substituting waste extensive goods for waste intensive goods. In reality, consumers have the possibility of two kinds of substitution, namely substitution within a sector and substitution between sectors. Substitution within a sector makes it possible to choose between two products that are basically the same except for waste intensity. Substituting between sectors would mean changing consumption patterns. For example, Linderhof *et al* (2001) show that households in Oostzaan not only bought more products containing less packaging, an example of substitution within a sector, but also began to use diaper services instead of disposable diapers, an example of substitution between sectors after the introduction of unit-based pricing. Waste prevention through substitution within a sector would add a certain degree of complexity to the model, as different products within the same sector and their associated 'waste intensity' would have to be explicitly modeled. We have chosen to include only the more straightforward channel of waste prevention through substitution between sectors. As a consequence, the possibility of waste prevention may, therefore, be underestimated.

The available incineration capacity in the model will be limited. The total available incineration capacity will be insufficient to fully treat all municipal and production waste. Firms and municipalities, however, will have the option of exporting combustible waste and waste to be landfilled. Thus the limitations of available capacity can be avoided and firms and municipalities will have the option to incinerate all their waste. Transporting costs and international prices of incineration will of course play a role in the decision whether or not to export waste.

### 4.2.3 Flat fee pricing of waste collection services

Most municipalities have chosen to charge a fixed amount of money for waste collection, the so-called flat fee. In a flat fee-pricing scheme, the amount of money paid for waste collection is independent of the quantity of waste actually generated. The perceived price for waste collection, in economic terms the marginal perceived costs of generating waste, equals zero in such a case. If the price of a good equals zero the equilibrium demand for that good can no longer be determined through the normal demand and supply functions. In the general equilibrium framework in particular, where it is assumed that some equilibrium price will ensure that demand equals supply, the zero price poses a problem. To implement a zero price in a general equilibrium model, we thus require an indirect approach. It is possible to implement a zero perceived price by using subsidies that compensate households for the cost of waste generation.

Households pay a fixed lump-sum transfer to the government for the collection of waste, based on the flat fee. This lump-sum transfer takes away part of the households' income. Therefore, the total expenditure of the households declines. The expenditure pattern, *i.e.* the percentage of income the households spend on a certain product will, however, not be affected.

In the model presented here, private households demand waste collection services and pay an equilibrium price for these services. To introduce the zero perceived price, the government reimburses these costs to the consumers in the form of a subsidy, which equals the equilibrium price for every unit of waste collection services exactly. Thus, the perceived price of waste collection for the households equals zero. If the revenue of the lump-sum transfer is lower than the amount spent on the subsidy, the government expenditure decreases (in this case there is a net subsidy on waste generation). If the revenue of the flat fee is higher than the total costs, government expenditure increases. The idea of the subsidy-*cum*-lump-sum transfer scheme is illustrated in Figure 4.2.

Section 4.2.6 shows how the subsidy-cum-tax scheme can be implemented in a general equilibrium model.

### 4.2.4 International trade in waste

The incineration capacity in the Netherlands is not sufficient to treat all combustible waste. This causes waste that could be incinerated to be landfilled. By opening the country borders for combustible waste the limits on incineration capacity are removed. Figure 4.3 shows how international trade of combustible waste can be introduced in a general equilibrium model.

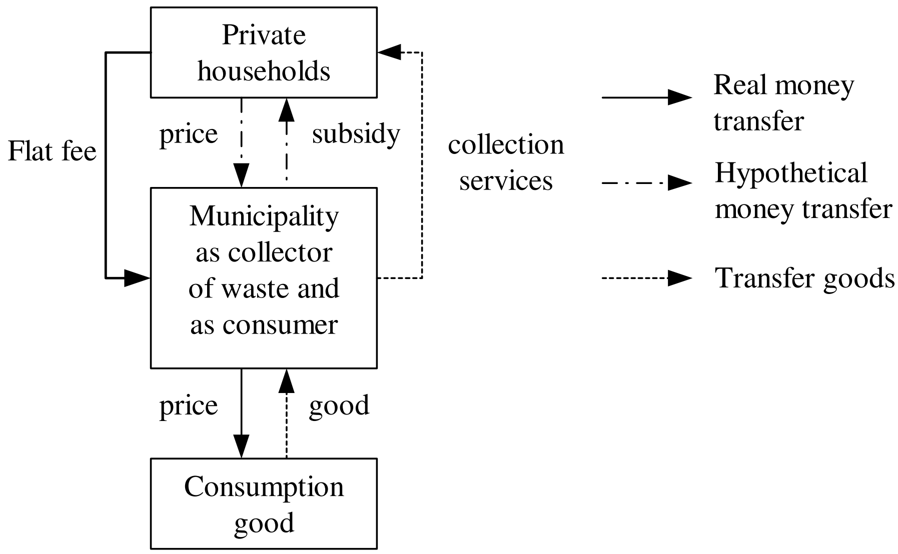


*Figure 4.2 The subsidy-cum-tax scheme.*

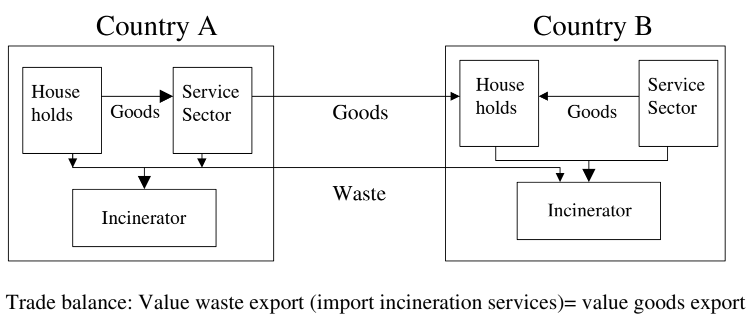


*Figure 4.3 Export of waste.*

Another country will be added to the model. This country consists of a consumer, a production sector, and an incineration plant. In this country consumers maximize utility by consuming the production goods just like in the original country. Country A will be an exporter of waste. This means that this country will import incineration services. The price of the imported incineration services is equal to the costs of producing these services and the transport cost. In a general equilibrium analysis, it is essential that the trade balance restriction is fulfilled. This means that the value of exported goods must be equal to the value of imported goods. Therefore the values of the imported waste incineration services must be offset by the value of exporting goods. Of course the same specification can be used to open the borders for all types of waste, including waste to be landfilled.

### 4.2.5 Model structure

#### Consumer utility function

In the Negishi format, total welfare is maximized subject to utility, balance, and production possibility constraints (Ginsburgh and Keyzer, 1997). The total welfare function is shown in equation 1.1. Total welfare (*TWF*) equals the sum of weighted utilities $(u_i)$ over consumer $i$ ( $i$=1,...,n).[19]

$$TWF(\alpha_i) = Max\sum_i \alpha_i u_i(x_{i,g}) \tag{4.1}$$

Consumers derive utility from the consumption of goods provided by the service sector $(x_i^g)$ where $g$= 'Wholesale market and auctions', 'Retail market', 'Repairment industry', 'Hotel and Catering industry', 'Transport services', 'Financial services', 'Other services' and 'Rest economy'. The utility of each consumer is weighted by a factor $\alpha_i$, the so-called Negishi weights[20].

Consumers generate waste by consuming products. Waste generation is dynamic; not all products will be transformed into waste immediately after consumption. Durable goods, for example, can continue to function properly for several years. If one looks at an infinite time scheme, every good will turn into waste. At any point in time, however, only part of the products will be transformed into waste. To include this dynamic aspect in a comparative static model, waste is determined as a fraction $\beta^g$ of the consumption product[21]. Total waste generation per consumer $(W_i)$ is equal to a fixed percentage of total consumption. The fraction of waste contained in a product differs for the three types of consumption goods. Agricultural and industrial goods are relatively waste intensive and thus $\beta$ will be positive for these goods; consumption of services does not generate waste and thus $\beta$ is equal to zero in this case. The government only consumes the goods 'rest economy' and we assume that the government does not generate waste; therefore, in the following equation a subset $c$ is used, which encompasses only the private households[22].

$$W_c = \sum_g \beta_g x_{c,g} \tag{4.2}$$

All waste that is generated has to be dealt with. Private households can chose to either recycle the waste by demanding waste recycling services $(x_{c,r})$ or to allow the waste to be collected by demanding waste collection services $(x_{c,w})$.

---

19 In the model application presented in the next Chapter we will distinguish only two consumers and the government, the model structure however is such that it is easy to distinguish many more consumers.

20 In the Negishi format, the equilibrium solution is found with the help of an iterative process. Given initial values for the Negishi-weights based on the income of a consumer, the model is solved and prices for each commodity are calculated as shadow prices. Subsequently, the budget constraint for each consumer is checked. If one or more consumers in the model spend more or less than their income, the Negishi weight for that consumer is adjusted. The model is then solved again with the adjusted Negishi-weights. The process continues until the budget constraints of all consumers hold. See for more information Negishi (1972) or Ginsburgh and Keyzer (1997).

21 Implicitly this means that part of the used material accumulates in a stock of durable goods. This stock is not constant, new materials enter the stock and other materials leave the stock as waste. Therefore, at any given moment in time the material inflow does not have to be equal to the material outflow in the model.

22 Because the focus of this reseach is on householdwaste and waste from the service sector, we decided to disregard the possibility of the government producing waste for the case of simplicity.

$$W_c = x_{c,r} + x_{c,r} \tag{4.3}$$

## Production functions

All production sectors can use two primary production factors, namely capital *(k)* and labour *(l)* and 5 intermediate inputs, namely ‘rest economy’ (*re*), virgin material ($m^v$), recycled material ($m^r$), incineration services ($w^i$) and landfilling services ($w^l$). All producers generate commodities $y_j$ within their given production set $Y_j$. The production set for the 7 consumption goods, is given by a nested Leontief-CES production function, which depends on the input of capital, labour, rest economy, virgin material, recycled material, and waste treatment services[23].

$$Y_j = A_j \left[ min \left\{ \begin{array}{l} CES\{CES(k_j, l_j; \sigma^{kl}), CES(w_j^{is}, w_j^{ls}; \sigma^{is,ls}); \sigma^{kl,wt}\}, \\ re_j, \\ CES(m_j^{v}, m_j^{r}; \sigma^{vr}) \end{array} \right\} \right] \tag{4.4}$$

$for\ j =$ service sector, rest economy

Where *A* stands for the technology level.

Producers can recycle waste by substituting capital and labour for waste collection services. The costs of recycling is determined by the relative ease of substitution, that is the substitution elasticity $\sigma^{kl,wt}$.

The production set for the producer of recycled material is given by a nested CES-function, which depends on the input of capital, labour, and recyclable waste:

$$Y_j = A_j \left[ CES\left\{ CES(k_j, l_j; \sigma^{kl}), X_r; \sigma^{pr} \right\} \right] \qquad for\ \text{j} = recycling\ \ services \tag{4.5}$$

Where $X_r$ is the total quantity of recyclable waste generated by the private households.

The production set for the producer of collection services is indicated by a nested Leontief-CES-function, which depends on the input of capital, labour, incineration services, and landfilling services:

$$Y_j = A_j \left[ min\left\{ CES(k_j, l_j; \sigma^{kl}), CES(w_j^{is}, w_j^{ls}; \sigma^{il}) \right\} \right] \quad for\ \text{j} = collection\ \ services \tag{4.6}$$

23 The notation $z$=*CES(x,y;σ)* reflects the following function: $z = \left( x^{\frac{(\sigma-1)}{\sigma}} + y^{\frac{(\sigma-1)}{\sigma}} \right)^{\frac{\sigma}{\sigma-1}}$ If a good is produced with production factors that are completely complementary (σ→∞), a Leontief production function can be used as a special case of the CES-production function. The standard notation for a Leontief production function is: *z=min(x,y).* A CES function can be nested. This means that, for example, the variable *x* in the equation above actually represents another function. In this chapter, several nested CES functions are used.

The production sets of all other production sectors are defined by CES-functions, which only depend on the input of capital and labour.

### Balance equations

As in any general equilibrium model, demand for commodities should be equal to the supply of these commodities. This is ensured by the following balance equations. First of all, total demand for consumption good *g* by consumer *i* and total demand for intermediate good *g* by producer *j* must not exceed the total supply ($y_g$) of good *g,* where *g* is an index of the six goods produced by the service sector: 'Wholesale market and auctions', 'Retail market', 'Repairment industry', 'Hotel and Catering industry', 'Financial services', 'other services' and the sector 'Rest economy'. The prices of the commodities can be determined from the balance equations by calculating the shadow price of the balance equation. In the following equations, this is symbolized by the '$\perp$' and a price variable *p*.

$$\sum_i x_{i,g} + \sum_j x_{j,g} \leq y_g \qquad \perp p_g \tag{4.7}$$

Total demand of all firms *j* for the intermediate goods: 'virgin material' ($m_j^v$), and 'recycled material' ($m_j^r$), must not exceed total supply of these materials ($y$). Since virgin materials and recycled materials are intermediate goods only, *i.e.* not demanded by the consumers, the only demand comes from firm *j*.

$$\sum_j m_j^v \leq y_v \qquad \perp p_v \tag{4.8}$$

$$\sum_j m_j^r \leq y_r \qquad \perp p_r \tag{4.9}$$

Total demand for the services: 'recycling services' ($x_{rs}$) and 'waste collection services' ($x_w$) by consumer *c* must be equal to or less than the total supply of these services.

$$\sum_c x_{c,rs} \leq y_{rs} \qquad \perp p_{rs} \tag{4.10}$$

$$\sum_c x_{c,w} \leq y_w \qquad \perp p_w \tag{4.11}$$

Total demand for the intermediate good: 'waste treatment service' ($w_{j,d}$), where d is a subset of j including incineration and landfilling services, must be equal to or less than total supply of these waste treatment services. To use the waste treatment services, the waste has to be transported to the waste treatment plant. These transport costs are represented by the transport matrix (*T*).

$$\sum_j w_{j,d} T_{j,d} \leq y_d \qquad \perp p_d \tag{4.12}$$

Total demand of primary factors must be equal to or less than total supply of these factors ($\overline{K}, \overline{L}$). The total supply of capital and labour is equal to the sum of initial supply of capital and labour of each consumer.

$$\sum_j k_j \leq \sum_i \overline{K_i} \qquad \perp p_k \tag{4.13}$$

$$\sum_j l_j \leq \sum_i \overline{L_i} \qquad \perp p_l \tag{4.14}$$

Prices for all commodities are calculated as the marginal value of the associated balance equations. The consumer obtains income by selling production factors, capital, and labour and spends his income on the six goods produced by the service sector, recycling services and waste collection services.

Incineration capacity in the Netherlands is limited. There is not enough incineration capacity available to incinerate all municipal solid waste and all waste of the service sector. To simulate this, we will put an upper limit on the available capital for the incineration sector.

$$k_{is} \leq \overline{K_{is}} \tag{4.15}$$

If the demand for incineration services is higher than the maximum available supply, the price of incineration will rise above the marginal production costs. The incineration sector will in this case make a profit. This profit is divided over the consumers and increases the income of the consumers.

The government derives its income from both a lump-sum transfer[24] paid by the consumers (*LS*). The government spends its income on the consumption of the 'rest economy' good. The value of government consumption is kept constant at its benchmark level. This means that the lump-sum transfer must be variable. If, for example, the income of the government increases due to an increase of the landfill tax, consumers will be reimbursed by the government through a decrease in the lump-sum transfer.

$$\sum_g p_g x_{c,g} + p_{rs} x_{c,rs} + p_w x_{c,w} + LS_c = p_k \overline{K_c} + p_l \overline{L_c} + profit_c \tag{4.16}$$

$$\sum_g p_g x_{gov,g} = \sum_c LS_c$$

### 4.2.6 Description of the model including a flat fee for waste collection and the landfill tax

Up till now the model described has no distortions or imperfections. Thus the equilibrium solution will be first-best. In this Chapter we will show how the market distortion flat fee pricing can be introduced. Due to this distortion, the solution found by the model including the flat fee can only be a second-best equilibrium.

To implement the subsidy-*cum*-tax scheme, as discussed in Section 4.2.3, a subsidy term is added to the objective function (equation 4.1)[25]. This subsidy term works like a benefit on the allocation of production output. Maximum social welfare now depends on the weighted utility of consumer *i* on the one hand and on the total benefits of the subsidy ($\xi^{cw} X_w$) on the other, where $X_w$ stands for the total quantity of waste generated and $\xi$ stands for the subsidy wedge, which is the total amount of money spent on the subsidy

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24 A lump-sum transfer will only affect the income level of the consumer and thus the total expenditure of that consumer. It will not result in a change of the consumption pattern of that consumer.

25 See Ginsburgh and Keyzer (1997) for details on this procedure.

per unit of waste. In the same way a tax on landfilling of waste can be included by adding a tax term ($\xi^{tl}TL$) to the welfare function (where *TL* is the total amount of waste landfilled and $\xi^{tl}$ stands for the tax wedge).

$$TWF(\alpha) = \max \sum_i \alpha_i u_i(x_{i,g}) + \xi^{xw} X_w + \xi^{tl} TL \tag{4.17}$$

$$x_{i,g} \geq 0\ ,\ \ w_i \geq\ 0,\ \ r_i\ \ \geq\ 0\ \ all\ i\ \ ,\ \ \ y_j\ \ all\ j$$

As shown by Ginsburgh and Keyzer, the only way to introduce subsidies and taxes in the Negishi format is by adding the subsidy/tax to the social welfare function. This is done solely to change the perceived price of waste collection. It does not imply that introducing subsidies would positively influence social welfare of a region. The social welfare calculated by this model is not comparable with the social welfare calculated by the model presented in Section 4.2.5. The presence of the subsidy in the welfare function is for technical reasons and specific to the Negishi format of the model. If the model were written in another format, the subsidy would not have been made explicit in the welfare function.

The subsidy wedge *($\xi^{v}$)* is defined as the difference between the equilibrium price for waste collection *($p_w$)* and the perceived price *($p_{c,w}$)*. In the present case, the perceived price of waste collection equals zero, thus the subsidy wedge is equal to the equilibrium price of waste collection. The tax wedge *($\xi^{tl}$)* is equal to the landfill tax[26].

The balance equation for waste collection services (equation 4.11) is rewritten as follows:

$$X_w \leq y_w \qquad\qquad \perp p_w \tag{4.18}$$

$$\sum_c x_{c,w} \leq X_w \qquad\qquad \perp p_{c,w} \tag{4.19}$$

In equation 1.17 the shadow price of waste collection has been calculated. This price equals the marginal production costs. In equation 4.18, the shadow price of waste collection, as consumers perceive it, is calculated. This price equals the equilibrium price minus the subsidy[27].

The balance equation for the demand of landfilling services (equation 4.12) is similarly rewritten as:

$$TL \leq y_{ls} \qquad\qquad \perp p_{ls} \tag{4.20}$$

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26 Just like the Negishi weights, finding the optimal subsidy and tax wedge is an iterative process. For each run op the model the subsidy and tax wedge are updated according to the prices calculated in the previous run. Only if the budget constraint of each consumer holds will the equilibrium subsidy and tax wedge be found.

27 Note that mathematically speaking, the introduction of the total waste demand variable is irrelevant. $X_w = \sum_c x_{c,w}$ can be substituted in the balance equation in the equilibrium solution. The distinction of $X_w$, however, enables the separation of the equilibrium price for waste collection and the perceived price.

$$\sum_j w_j^{ls} \leq TL \qquad \perp p_{tax,ls} \tag{4.21}$$

The new budget constraint for the private households is defined as follows:

$$\sum_g p_g x_{c,g} + p_{rs} x_{c,rs} + p_{c,w} x_{c,w} + F_c + LS_c = p_k \overline{K}_c + p_l \overline{L}_c + profit_c \tag{4.22}$$

Private households spend their income on the consumption of consumer goods, recycling services and collection services (bear in mind that $p_{c,w}$ is zero, so the costs of consumption of waste collection services are equal to zero) and pay a flat fee *(F)* to the government for the collection of waste.

The new budget constraint of the government is defined as follows:

$$\sum_g p_g x_{gov,g} + S = \sum_c LS_c + \sum_c F_c + \sum_j TAX_j \tag{4.23}$$

The government spends its income on consumer goods and the subsidy costs *(S)*. Since the government does not generate waste, it does not need to spend any income on the collection of waste. We assume that the government owns primary factors and earns income both from selling these primary factors, benefits of the flat fee and income from the landfill tax (TAX).

The size of the subsidy costs / tax benefits depends on the total amount spent on the subsidy for waste collection / waste landfilled, which is calculated as follows:

$$S = \xi^{xw} \sum_c x_{c,w} \qquad \mathrm{TAX} = \xi^{lw} \sum_j w_{j,ls} \tag{4.24}$$

The total transfer equals the subsidy wedge *($\xi$)* multiplied by the total demand for waste collection services. The subsidy /tax wedge is calculated as follows:

$$\xi^{xw} = p_w - p_{c,w} \qquad \xi^{lw} = p_{ls} - p_{tax,ls} \tag{4.25}$$

The subsidy wedge is equal to the real price of waste collection minus the perceived price of waste collection.

### 4.2.7 Description of model including international trading of waste

International trade is added by adding another country, which produces waste treatment services and a consumption good. In this simplified foreign country we distinguish 1 consumer who consumes the consumption good ‘rest economy’. The social welfare function is now calculated by maximizing the sum of the utility of consumers *i* in country *n.*

$$TWF(\alpha) = \max \sum_i \sum_n \alpha_{i,n} u_{i,n}(x_{i,g,n}) + \xi^{xw} X_w + \xi^{tl} TL \tag{4.26}$$

$$x_{i,g,n} \geq 0\,,\; w_{i,n} \geq 0,\; r_{i,n} \geq 0 \;\; all\; i,\; n \;\;,\;\; y_j \;\; all\; j$$

Countries can trade both incineration services and the consumption good ‘rest economy’[28]. The balance equation for these sectors changes to:

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[28] In one of the scenarios presented in the next chapter, trade in landfill waste is also allowed. The model will be adjusted accordingly.

$$\sum_j (w_{j,n}^{is} T_{j,n}) + z_{is,n} T_{z,n} \leq y_{is,n} \qquad \perp p_{is,n} \textit{ for each country n} \qquad (4.27)$$

$$\sum_i (x_{i,rest,n}) + z_{rest,n} \leq y_{rest,n} \qquad \perp p_{rest,n} \textit{ for each country n} \qquad (4.28)$$

Where z is the net trade, note that if z is negative the country is a net importer of that good and if z is positive the country is a net exporter of that good. The trade balance needs to be added to the model, which insures that the value of exported goods is equal to the value of imported goods.

$$\sum_j p_{j,n} z_{j,n} = 0 \qquad \textit{for each country n} \qquad (4.29)$$

## 4.3 Empirical analysis: effectiveness of the landfill tax

### 4.3.1 Data description

The model presented in the previous Chapter is used to evaluate the effectiveness of the landfill tax in the Netherlands. We use data for the Netherlands from the year 2002 in this evaluation. The data is gathered from CBS and the waste management council (AOO).

### The benchmark case

The social accounting matrix of the economy is presented in Table 4.1. Supply or producers' output, capital and labour are given as positive values; demand or producer inputs and consumption are given as negative values[29]. To keep the model as simple as possible, government income is dependent on a lump-sum transfer instead of an income from taxes on labour and consumer goods. This has been added to the social accounting matrix.

Since the service sector only makes up a small part of our total economy, it is unlikely that changes in demand for the intermediate good 'rest economy' and 'virgin material' will have a large impact on the price of these goods. Therefore we assume that the government consumes a large part of the good 'rest economy' and that a large part of the available virgin material is used in the production of the good 'rest economy'.

[29] The entries in the column times the corresponding prices add up to zero to ensure that the zero profit condition holds: value of inputs equals value of outputs. The entries in the column of each consumer times the corresponding price add up to zero to ensure that the budget constraint holds: each consumer spends exactly its income on the consumption of goods and services. The entries in each row times the corresponding prices add up to zero to ensure that each market clears: total demand for each commodity must equal total supply. In Table 4.1 the rows and columns may not add up to zero exactly due to the rounding off of several numbers.

*Waste generation*

In total, consumers and the service sector generated 6121 ktonne of rest waste in 2002. About 80% of the rest waste stream was incinerated, the rest landfilled. We assume that all waste collected separately is recycled. The service sector in total separated about 1670 ktonne of waste. Municipalities collected 4439 ktonne of separated waste.

Consumers are divided into 2 types. The first type lives in a municipality that charges a flat fee for the collection of waste, the second type lives in a municipality that charges a unit-based price for the collection of waste. The share between consumer 1 and consumer 2 is determined based on the number of households living in 'flat fee municipalities' and the number of households living in 'unit-based pricing municipalities'. Data from AOO shows that consumers living in 'unit-based pricing municipalities' generate less rest waste and more recyclable waste than households living in 'flat fee pricing municipalities'.

Consumers that pay a fixed amount of money for the collection of rest waste, the so-called flat fee, face a zero marginal price for waste generation. This is modeled as if these consumers pay the equilibrium price for waste collection (0.357 million euro per ktonne) however, the government reimburses these costs to the consumers in the form of a subsidy, which is exactly equal to the equilibrium price for every unit of waste collection services. Thus, the marginal price of waste generation for the households equals zero. The consumers pay a total amount of 1343 million Euros for the collection of waste. On average, the fee paid by the consumers covers only 95% of the real costs (AOO, 2002a). This means that the real costs of waste collection and thus the amount spent on the subsidy on waste collection equals roughly 1414 million Euros.

The two consumer types only differ in the price they face for the collection of waste. Their consumption preferences are exactly the same.

*Prices and landfill tax*

Data from AOO (2003) shows that the consumers would pay on average 357 euro per tonne of waste (0.357 million per ktonne), given a 100% cost-coverage rate. This price is shown in table 4.1 in the price column. Because prices of recycling and recycled material proved impossible to find we need to establish them with a kind of modeling trick. We know that consumer 2 faces a unit-based price for the collection of waste. If recycling would be a cheaper option than demanding collection services, it can be expected that this consumer would recycle more. Therefore we can assume in the benchmark case that the price of recycling and therefore also the price of recycled waste is the same as the price of waste collection[30]. The prices of all other products have been normalized to 1, according to the Harberger convention[31].

[30] Note that this deduction of recycling prices is only possible if we assume that consumers have full information and base their decision to recycle solely on differences in price level of waste collection and recycling.

[31] Following standard practice, we adopted the Harberger convention in the benchmark data for all unknown prices. The Harberger convention consists of normalizing prices to unity. Quantities in the benchmark data represent expenditures, or how much of that good or factor one can buy for €1. It should be noted that an Arrow-Debreu economy only depends upon relative prices. Doubling all prices doubles both money profits and income, which results in the same equilibrium outcome.

*Table 4.1 Benchmark data input-output model (waste sectors in ktonne, other sectors in million euro).*

| | Wholesale | Retail | Catering | Repair-ment | Transport sector | Financial | Other | Rest economy | Virgin material | Recycled material | Recycling Services | Collection waste | Landfill | Incin-era-tion | Consumer 1 | Consumer 2 | Govern-ment | Price |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wholesale sector | 18136 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -14872 | -3264 | 0 | 1,000 |
| Retail sector | 0 | 11010 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -9028 | -1982 | 0 | 1,000 |
| Catering sector | 0 | 0 | 3679 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -3017 | -662 | 0 | 1,000 |
| Repairment sector | 0 | 0 | 0 | 5238 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -4295 | -943 | 0 | 1,000 |
| Transport sector | 0 | 0 | 0 | 0 | 34376 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -28188 | -6188 | 0 | 1,000 |
| Financial sector | 0 | 0 | 0 | 0 | 0 | 32965 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -27032 | -5934 | 0 | 1,000 |
| Other sector | 0 | 0 | 0 | 0 | 0 | 0 | 5859 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -4804 | -1055 | 0 | 1,000 |
| Rest economy | -77 | -86 | -47 | -27 | -34 | -82 | -77 | 10429 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -10000 | 1,000 |
| Virgin material | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -5000 | 5000 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1,000 |
| Recycled material | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -2642 | 0 | 2642 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1,000 |
| Recycling Services | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4439 | 0 | 0 | 0 | -3464 | -975 | 0 | 0,357 |
| Recycled waste | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -4439 | 4439 | 0 | 0 | 0 | 0 | 0 | 0 | 0,357 |
| Collection waste | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4614 | 0 | 0 | -3959 | -654 | 0 | 0,357 |
| Landfill | -131 | -143 | -109 | -49 | -53 | -134 | -160 | 0 | 0 | 0 | 0 | -692 | 1470 | 0 | 0 | 0 | 0 | 0,049 |
| Incineration | -205 | -224 | -171 | -78 | -84 | -210 | -251 | 0 | 0 | 0 | 0 | -3922 | 0 | 5145 | 0 | 0 | 0 | 0,117 |
| Capital | -2817 | -2386 | -715 | -1636 | -18730 | -4585 | -1060 | -787 | -4750 | -266 | -797 | -447 | -35 | -546 | 32441 | 7137 | 0 | 1,000 |
| Labour | -15199 | -8490 | -2879 | -3559 | -15593 | -28253 | -4668 | -2000 | -250 | -791 | -2373 | -670 | -43 | -61 | 69536 | 15264 | 0 | 1,000 |
| Transport cost | -3 | -4 | -3 | -1 | -1 | -3 | -4 | 0 | 0 | 0 | 0 | -47 | 13 | 53 | 0 | 0 | 0 | 1,000 |
| Fee | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -1343 | 0 | 1343 | 1,000 |
| Subsidy | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1414 | 0 | -1414 | 1,000 |
| Tax | -11 | -11 | -9 | -4 | -4 | -11 | -13 | 0 | 0 | 0 | 0 | -46 | 0 | 0 | 0 | 0 | 118 | 1,000 |
| Lumpsum | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -8161 | -1791 | 9952 | 1,000 |

*Note: ‘Fee’ is the flat fee consumers pay to the government for collection of waste, ‘Subsidy’ stands for the total amount of money the government gives for collection of waste as a subsidy to the consumers. ‘Lumpsum’ stands for a lumpsum transfer from the consumers to the government. The price column gives the prices of all commodities.*

*Production*

Data about production costs of the seven service sectors is gathered from CBS (Statline). The value of production shown in table 4.1 is the value of production sold to households. Intermediate deliveries between service sectors are not taken into account in this model. The intermediate deliveries from the sector 'rest economy' can be seen as material inputs into the production. The total amount of waste generated by the sector is directly linked to the amount of material input into the production process.

Recycling by the different service sectors is modeled as a substitution between capital and demand for waste treatment services. Recycled waste is directly reused by the production sector. Part of the benchmark capital demand is reserved for recycling. Table 4.2 shows the benchmark recycling data

*Table 4.2 Service sector recycling.*

| Service sector | Recycling (kton) |
|---|---|
| Wholesale | 333 |
| Retail | 385 |
| Catering | 137 |
| Repairment | 111 |
| Transport sector | 162 |
| Financial | 376 |
| Other | 274 |

*Transport costs and economies of scale*

The transport costs of transporting a tonne of waste to a waste treatment unit depends on the distance traveled. It is here assumed that a larger unit will be located further from the municipality than a smaller unit. In Table 4.3 the transport distances are presented. These distances are based on the average distances from a municipality to an incinerator or a landfill site in the Netherlands (AOO, 2003d).

*Table 4.3 Transport distances.*

| Size of waste treatment unit | Distance from municipality (in km's) |
|---|---|
| Incinerator | 40 |
| Landfill | 35 |

*Substitution elasticities*

The amount of recycling by the service sector is determined by the substitution elasticity between capital and labour on the one hand and waste treatment services on the other hand. Based on information about the actual amount of waste recycled and the costs of waste treatment in the period 1995-2003, we estimated the substitution elasticities for the seven service sectors. These substitution elasticities are shown in table 4.4. Higher substitution elasticities mean that these sectors are more sensitive and responsive to price changes. For example the transport sector has increased waste separation and recycling relatively more than the catering sector over the past 10 years. Given the same price change of landfilling for these two sectors, we can conclude that the transport sector is somewhat more sensitive to price changes and therefore has a slightly larger substitution elasticity.

*Table 4.4 Substitution elasticities.*

| | substitution elasticity waste treatment/ recycling ($\sigma^{kl,wt}$) |
|---|---|
| Wholesale sector | 0.37 |
| Retail sector | 0.38 |
| Catering sector | 0.29 |
| Repairment sector | 0.37 |
| Transport sector | 0.43 |
| Financial sector | 0.42 |
| Other sector | 0.31 |

Substitution elasticities between labour and capital for the different production sectors, between recycled material and virgin material, and between landfilling and incineration are given in Table 4.5. The production sectors use capital and labour as inputs for production. They can substitute between the use of capital and labour. Based on Draper and Manders (1996), we choose a substitution elasticity of 0.8.

*Table 4.5 Other substitution elasticities.*

| | production sectors | municipality |
|---|---|---|
| Sub.elas. labour & capital ($\sigma^{kl}$) | 0.8 | 0.8 |
| Sub.elas. recycled material & virgin material ($\sigma^{vr}$) | 2 | |
| Sub.elas. landfilling & incineration ($\sigma^{il}$) | 2 | 2 |

*Limitation incineration capacity*

To simulate the limited capacity of incineration plants, we put an upper limit on the available capital for incineration. This upper limit on capital use is equal to 1.1 times the available capital in the benchmark data set. Due to longstanding contracts between municipalities and incineration plants, we can safely assume that incineration plants will first treat municipal solid waste and will only treat waste from other sources if there is still capacity left. To simulate this, we will keep the price of incineration for municipalities equal to the benchmark level. The service sector will compete for the limited capacity that is left and price of incineration for the service sector will increase if the demand for these services increases.

### 4.3.2 Description of the scenarios

#### 1. Benchmark scenario

Based on the benchmark data described in the previous Chapter we will determine how effective the landfill tax is. We will vary the landfill tax from very low to very high and show how different variables are affected. We will look specifically at the variables:

- Amount of waste landfilled;
- Amount of waste incinerated;
- Amount of waste recycled;
- Welfare economy (in terms of total consumption).

### Other scenarios

The effectiveness of the landfill tax will be influenced by several characteristics of the waste market. In three scenarios we will analyze how these characteristics influence the results by comparing these results to the benchmark scenario.

### 2. Flat fee pricing waste collection

In the benchmark scenario we based the share of 'flat fee municipalities' and 'unit-based pricing municipalities' on the actual number of households living in 'flat fee municipalities' and 'unit-based pricing municipalities' in 2002. In this scenario we will vary this share from hardly no 'flat fee municipalities' to hardly no 'unit-based pricing municipalities' and analyze whether a changing landfill tax will affect the amount of recycled waste, landfilled waste and incinerated waste considerably compared to the benchmark scenario.

### 3. International trade in combustible waste

In the third scenario we will introduce the possibility of international trade in combustible waste. By allowing export of combustible waste we will release the assumption of limited incineration capacity and we will analyze whether the behavior of the service sector is noticeably different if they have the option of increasing the amount of waste they incinerate. In this scenario we will assume that the amount of waste exported is not sufficient to change the price in the importing country[32].

### 4. International trade in both combustible waste and waste to be landfilled

In the fourth scenario we will also allow the possibility of international trade in waste to be landfilled. In contrast to the previous scenario we will assume that the export of waste is large enough to change prices of incineration and more importantly landfilling in the importing country.

### 4.3.3 Results of the ex-ante evaluation: benchmark scenario

By using the ex-ante model we will try to predict how industries and households react to a raise of the landfill tax. Therefore, we will vary the landfill tax from a very low level to a very high level and we will analyze how much waste will be landfilled, incinerated, and recycled.

In the ex-ante model, we do not take into account the external cost of waste treatment. Therefore, the introduction of a landfill tax will always result in a lower social welfare (measured in total utility from consumption) since a landfill tax will raise prices of production goods and services. We will compare the benefits of a landfill tax, measured in tonnes of waste recycled or incinerated instead of landfilled, against the social costs of the landfill tax measured in loss of consumption.

Figure 4.4 shows how the total amount of waste landfilled or incinerated is affected by the landfill tax rate. The marginal benefits of the landfill tax are especially high in the beginning. Going from a low landfill tax rate to a slightly higher one results in a big decrease in

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[32] Note that this scenario is basically the same as introducing sufficient incineration capacity.

waste landfilled. The amount of waste incinerated does not increase as much as the amount of waste landfilled decreases, indicating that households and industries do increase recycling. The current tax rate on landfilling (2005) amounts to 84.78 Euro per tonne. Due to the limited capacity of incineration plants, incineration can not increase after a certain level. Given a landfill tax of about 100 euro a ton, the complete incineration capacity is filled and therefore incineration can not increase anymore.

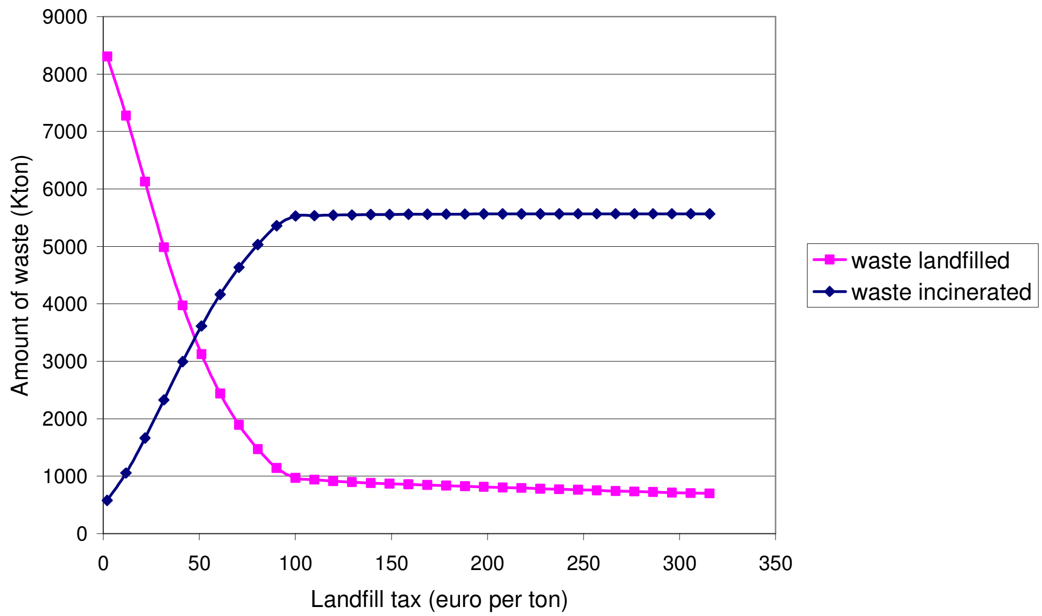


*Figure 4.4 Waste treatment and the landfill tax.*

Figure 4.5 shows the amount of waste treated for households and industries individually. The difference between household behavior and industries is quite noticeable. Municipalities, who collect household waste, switch from landfilling to incineration. If the landfill tax is very high, almost all household waste will be incinerated. The service sector also increase incineration and decreases the amount of waste landfilled. However, figure 4.5 shows that the amount of waste landfilled increases again when the tax rises above 100 euro per ton. This is caused by the limited capacity of incineration plants. Municipalities offer such a large quantity of waste to the incineration plants that the remaining capacity is to small to incinerate all the waste offered to it by the service sector. As a result, the service sector is forced to start landfilling the waste again.

Since industries hardly have the option of switching to incinerating waste instead of landfilling, due to the restriction put upon the available incineration capacity, it can be expected that industries will start to recycle more waste. The amount of waste recycled by the individual industries in the model is shown in Figure 4.6. Especially the sector 'transport sector' increases recycling a lot.

Figure 4.6 also shows that the service sector especially increases recycling when the landfill tax rises above 100 euro per ton. As explained above, this is due to the limited capacity of incineration plants, making it very costly to incinerate service sector waste.

Figure 4.7 shows the amount of waste generated and the amount of waste recycled in municipalities. We distinguish between municipalities that introduced unit-based pricing for waste collection and municipalities that still ask a flat fee for waste collection.

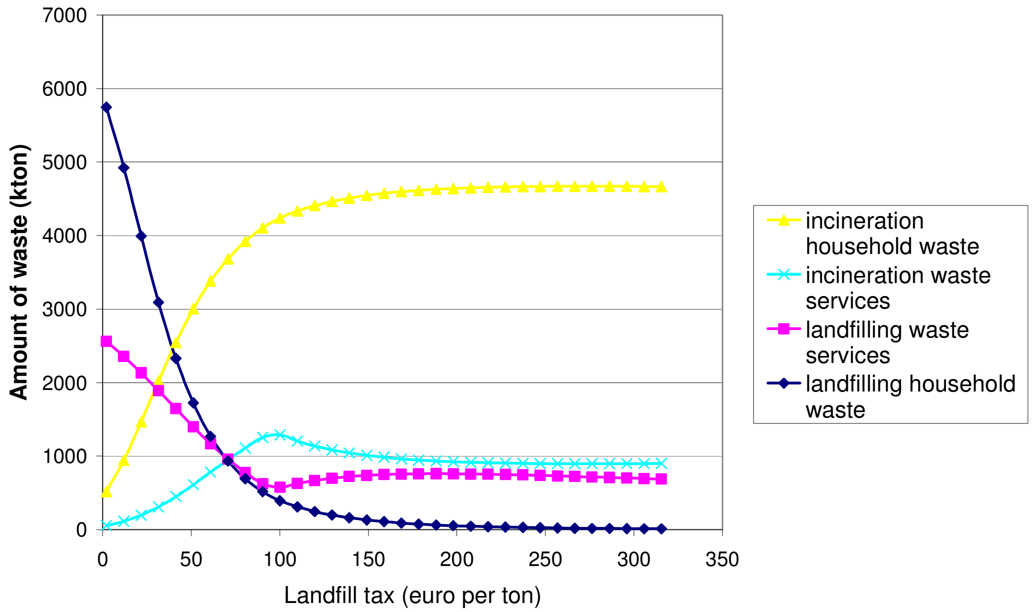


*Figure 4.5 Landfilling and incineration of household waste and waste from industries.*

In the municipalities that introduced a unit-based pricing system for waste collection, we see that households are affected by a higher landfill tax. Since the price they pay for waste collection will increase due to the higher landfill tax, they will increase their recycling efforts. Households living in a municipality that charges a flat fee for waste collection are not affected by the landfill tax.

Please note that we assume a benchmark level of recycling in both types of households. Data shows that households do recycle some of their waste independent of the costs of waste collection. It is clear that other concerns like environmental responsibility play a role in the decision whether or not a household will recycle. Recycling levels in a unit-based price municipality are generally higher than recycling levels in a flat fee municipality. Only if the landfill tax is between 0-20 euro per tonne, will the price incentive be so small that the recycling levels of both municipalities are comparable.

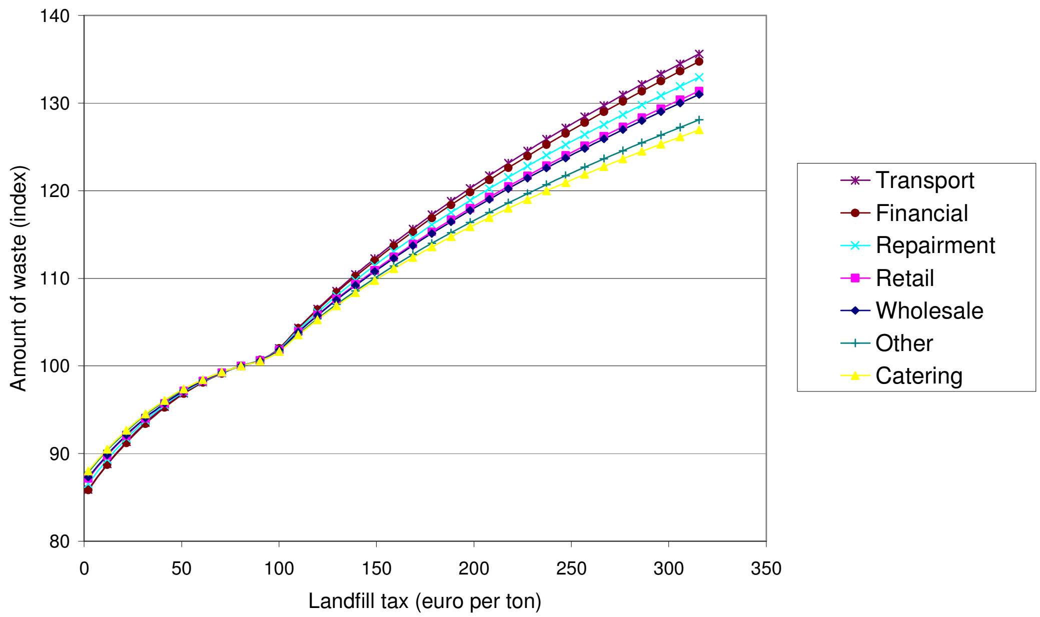


*Figure 4.6 Recycling by different industries(index current tax=100).*

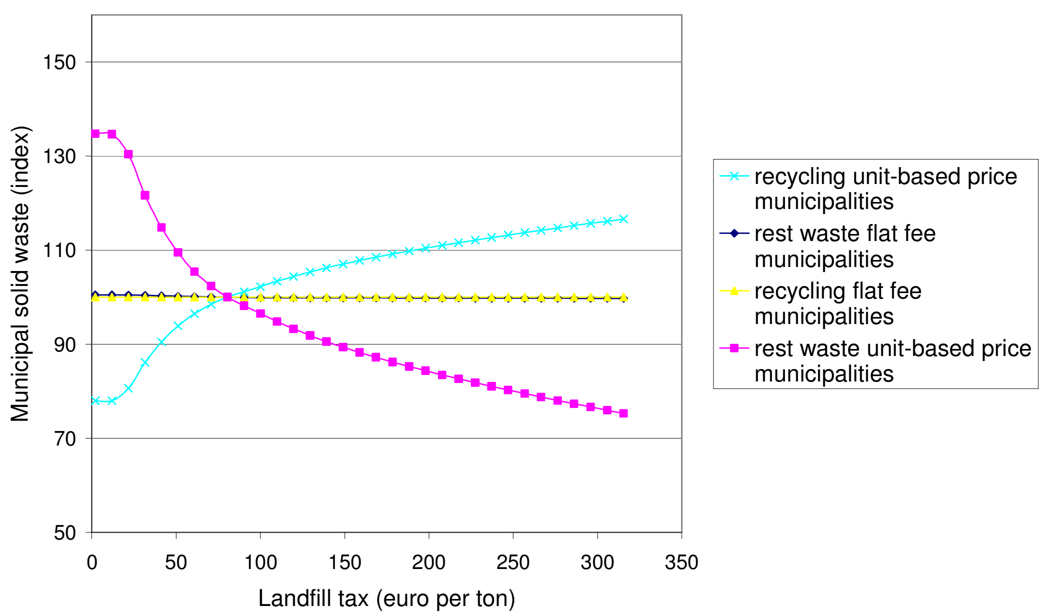


*Figure 4.7 Waste generation and recycling in municipalities (index current tax=100.).*

Figure 4.8 shows the loss of welfare due to the introduction of a landfill tax. The higher the landfill tax, the higher the loss of welfare in society.

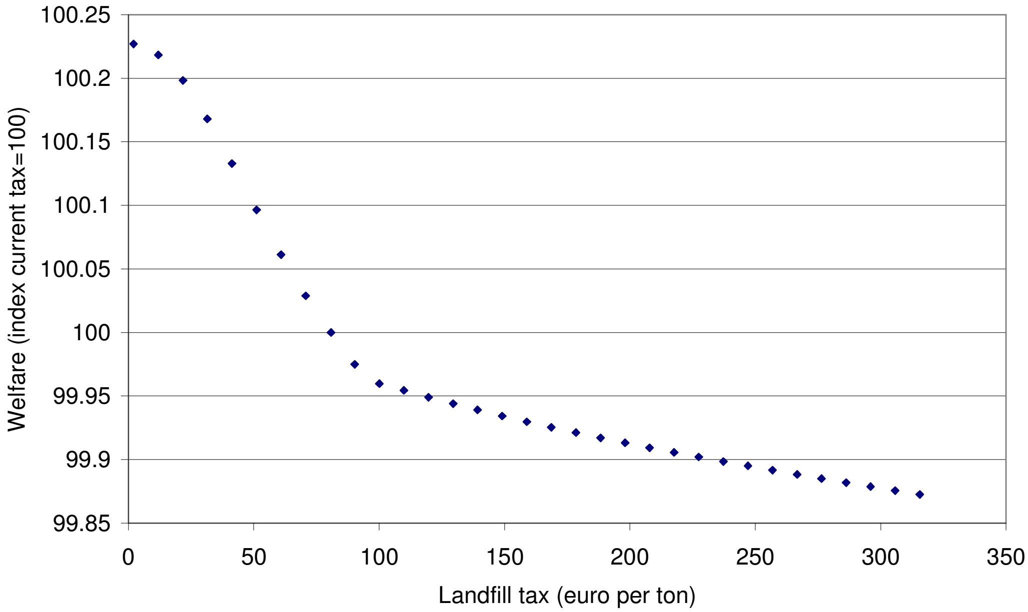


*Figure 4.8 Social welfare and the landfill tax (index current tax=100).*

Finally we make an analysis of the impact of the landfill tax. In order to compare the decrease in waste landfilled to the decrease in welfare, we divide the marginal decrease in waste landfilled by the marginal decrease in welfare of society due to an increase in the landfill tax. The impact is shown in Figure 4.9.

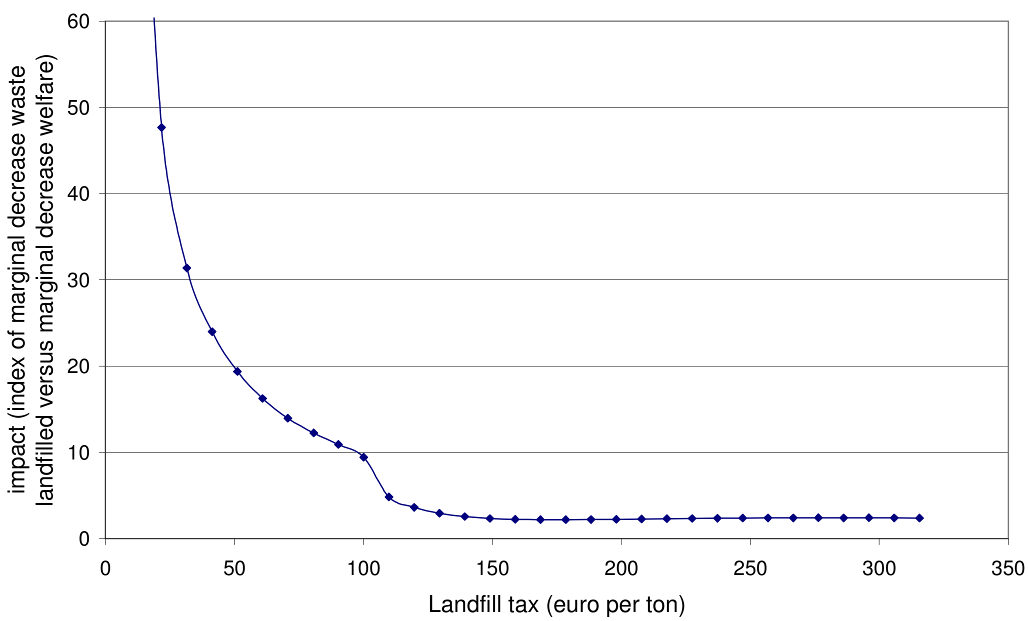


*Figure 4.9 Impact of the landfill tax.*

Going from a low to a slightly higher tax has a large impact on the amount of waste landfilled but not such a large impact on the welfare of the society (measured in consumption). However this impact rapidly decreases as the landfill tax increases. There is a discontinuity in the impact function around a landfill tax of 100 euro/ton. This is caused by the limited incineration capacity. Around a landfill tax of 100 euro per tonne all available incineration capacity is used and therefore the landfill tax cannot induce the service sector to substitute landfilling for incineration.

### 4.3.4 Results of the other scenarios

#### Scenario 2: Effect of unit-based pricing

In the second scenario we will investigate how the flat fee pricing scheme for collection of municipal solid waste influences the effectiveness of the landfill tax. In Section 4.3.3 we showed that households living in a municipality that charges a flat fee for the collection of waste, are not affected by the landfill tax. They will not increase recycling or waste separation because there is no direct link between waste generation and the costs of waste collection. Municipalities themselves, however, are affected by the landfill tax regardless of the price they charge for the collection of municipal solid waste. They will increase the quantity of waste incinerated and decrease the quantity of waste landfilled due to a raise in the landfill tax.

In this scenario we will compare two situations:

1. Full unit-based price: All municipalities charge a unit-based price for waste collection;
2. Partial unit-based price: 18% of the municipalities charge a unit-based price for waste collection, this is similar to the benchmark scenario as presented in the previous Chapter.

In Figure 4.10 we show the effect that a raise in the landfill tax will have on the total amount of municipal solid waste recycled. We only show in this figure how the quantity of waste recycled will increase if the landfill tax is raised from its present value (of about 80 euro). In Figure 4.10 we will compare the total amounts of waste recycled and waste offered as rest waste in the benchmark scenario (partial unit-base price) to the amounts of waste recycled and offered as rest waste in the new scenario (full unit-based price).

In the full unit-based pricing scenario, more waste is recycled. However, in both cases the increase in recycling is not very large. If we go from a landfill tax rate of about 80 euro to a rate of 300 euro, the quantity of waste recycled will increase with about 4% in the full unit-based price scenario and with about 2% in the partial unit-based pricing scenario.

#### Scenario 3: International trade in combustible waste

In the third scenario we will show how the limited capacity of incineration plants influences the effectiveness of the landfill tax. By allowing export of combustible waste to Germany, we can increase the incineration capacity. Incineration prices in Germany vary between 66 and 340 euro per tonne of waste (Landtag von Baden-Württemberg, 2002). In this scenario we will not include export of combustible waste to Belgium as there is no capacity available for foreign waste. In the next scenario we will allow some export of waste to Belgium.

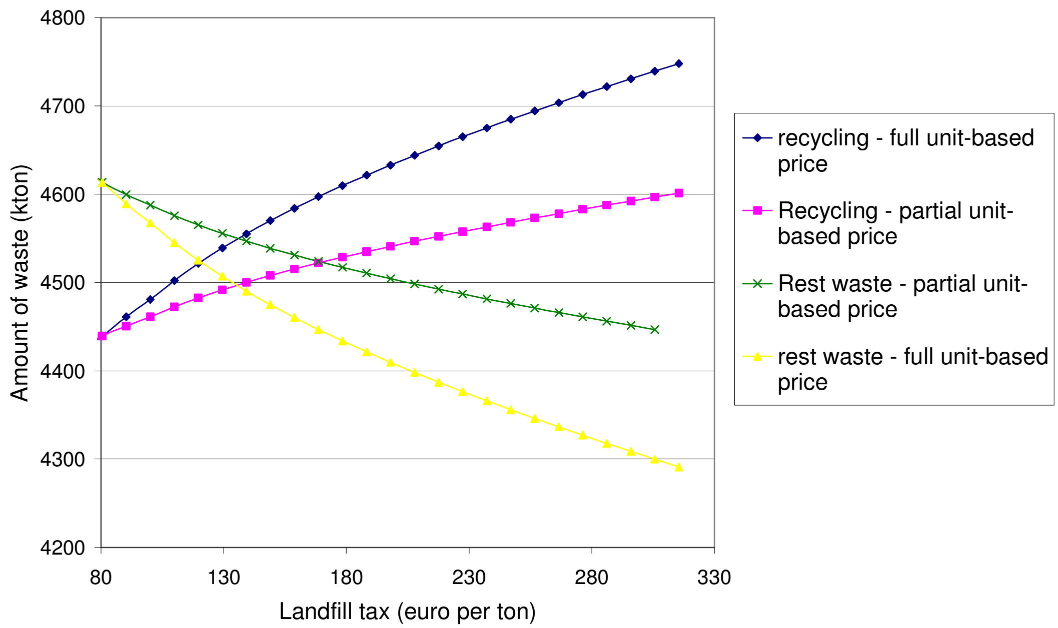


*Figure 4.10 Recycling and rest waste in scenario 1: effect of unit-based pricing.*

Note: recycling and waste production in the partial unit-based price are similar to the benchmark scenario.

Considering the large range of incineration prices in Germany, it is difficult to determine just what the average price of incineration of foreign waste in Germany would be. We will assume in this scenario that incineration on average is priced the same in Germany as in the Netherlands. The only price difference is caused by the transportation costs to the incineration plant. This of course is probably not completely realistic: if the price of incineration in Germany is higher than the price of incineration in The Netherlands we may overestimate the effects of opening the borders for combustible waste somewhat. However, the trend that we will show in the results in the next paragraph will not change. It only may take a slightly higher landfill tax to stimulate firms to export their waste instead of landfilling it. The differences in waste landfilled, incinerated, or recycled between the scenario where export is allowed and the benchmark scenario where export is not allowed is shown in Figure 4.11.

Due to the possibility of export of combustible waste we see that far more waste is incinerated. Landfilling decreases with nearly 80% if the landfill tax is increased to 300 euro. In contrast, if export is not allowed, landfilling hardly decreases due to the limited capacity of incineration plants. However, the option of export influences not only the choice between incineration and landfilling but also the choice between incineration and recycling. Since incineration becomes more readily available, we see that recycling declines a bit. Without export, the service sector will increase recycling with about 18% if the landfill tax is increased to 300 euro. With the possibility of export, recycling increases with about 11% given a landfill tax of 300 euro.

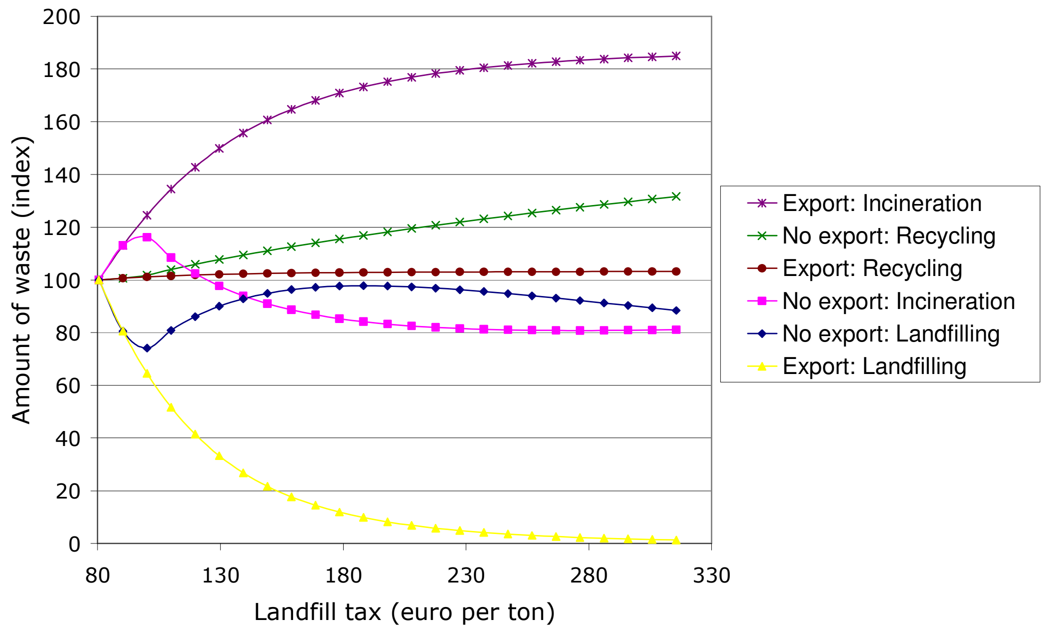


*Figure 4.11 Impact of export of combustible waste: results for the Service sector.*

Note: the results for recycling and incineration in the no export scenario are similar to the benchmark scenario.

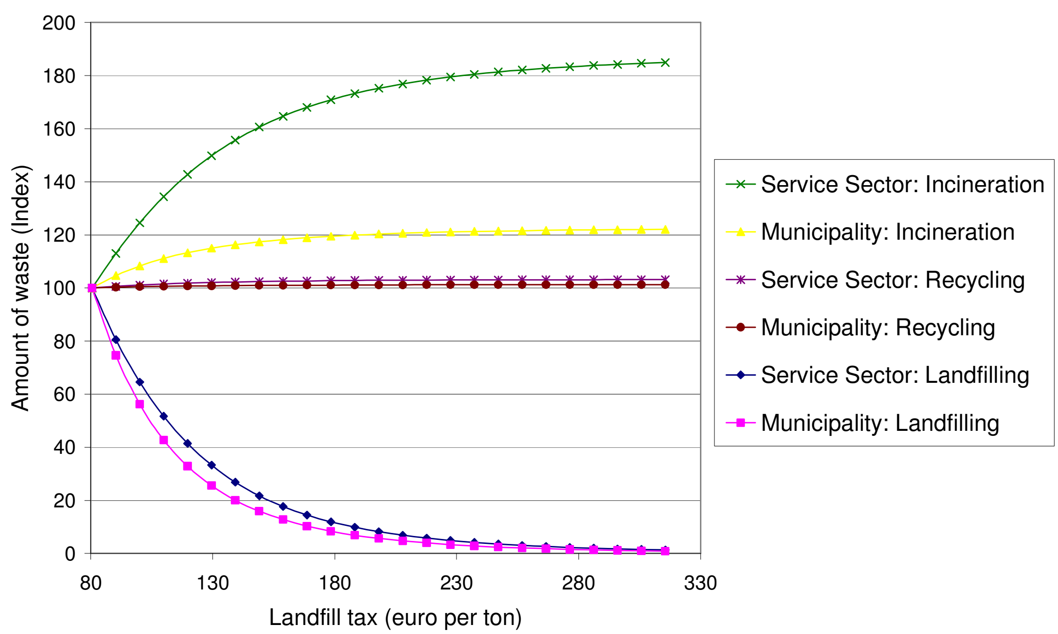


*Figure 4.12 Impact of export of combustible waste: results for the Service sector compared to the results of the households (index current tax=100).*

Figure 4.13 shows that all our landfill waste will be exported when the price of landfilling including the landfill tax exceeds the price of landfilling in Belgium and Germany. Since landfilling in either Belgium and Germany is less expensive than incineration in the Netherlands (even given transportation costs) there will be no substitution between landfilling and incineration anymore. The service sector will also not increase their recycling efforts anymore as shown in Figure 4.14.

Landfilling in Belgium is less expensive than landfilling in Germany. However, the capacity of the landfill sites in Belgium is limited. Therefore only part of the waste is exported to Belgium. The rest is exported to the more expensive landfill sites in Germany.

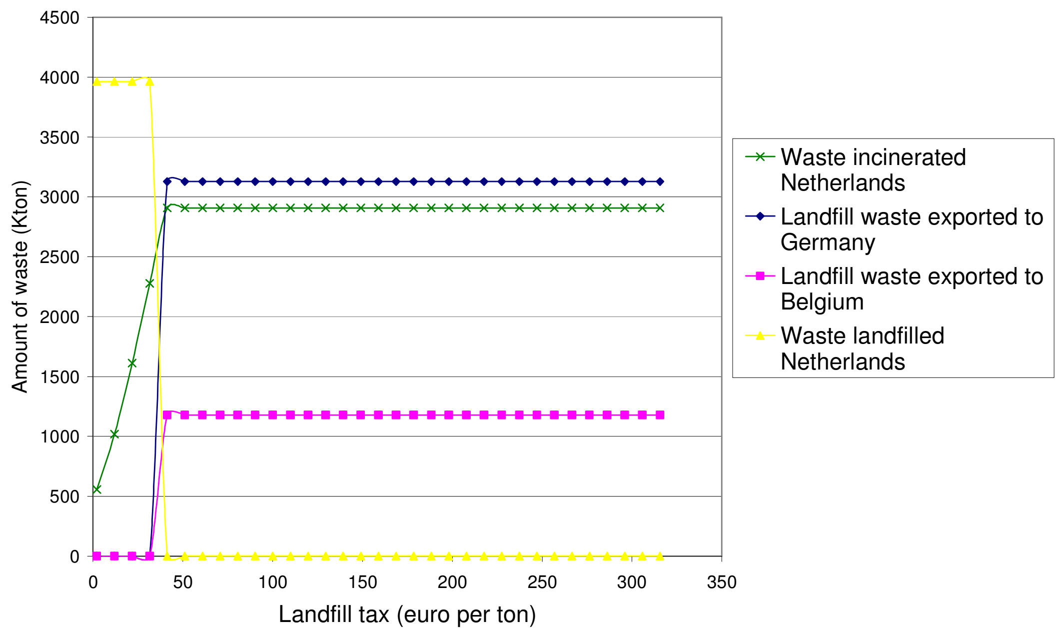


*Figure 4.13 Results scenario 4: export of waste to be landfilled.*

## 4.4 Landfill tax versus landfill ban

In 1995, the government in the Netherlands introduced a landfill ban, quickly followed by a landfill tax. In principle, the landfill ban means that it is prohibited to landfill recyclable or combustible waste in the Netherlands. Only if waste cannot be reused and the capacity to incinerate the waste is not sufficient, it is allowed to landfill recyclable and combustible waste. Up to 2002, due to the lack of capacity of waste incineration plants, industries and municipalities could relatively easily obtain exemptions from the landfill ban, especially in the early years following the introduction of the landfill ban. In 2002, there was a noticeable decrease in the amount of combustible waste landfilled, as can be seen in figure 4.15. Especially municipalities greatly decreased the amount of waste landfilled.

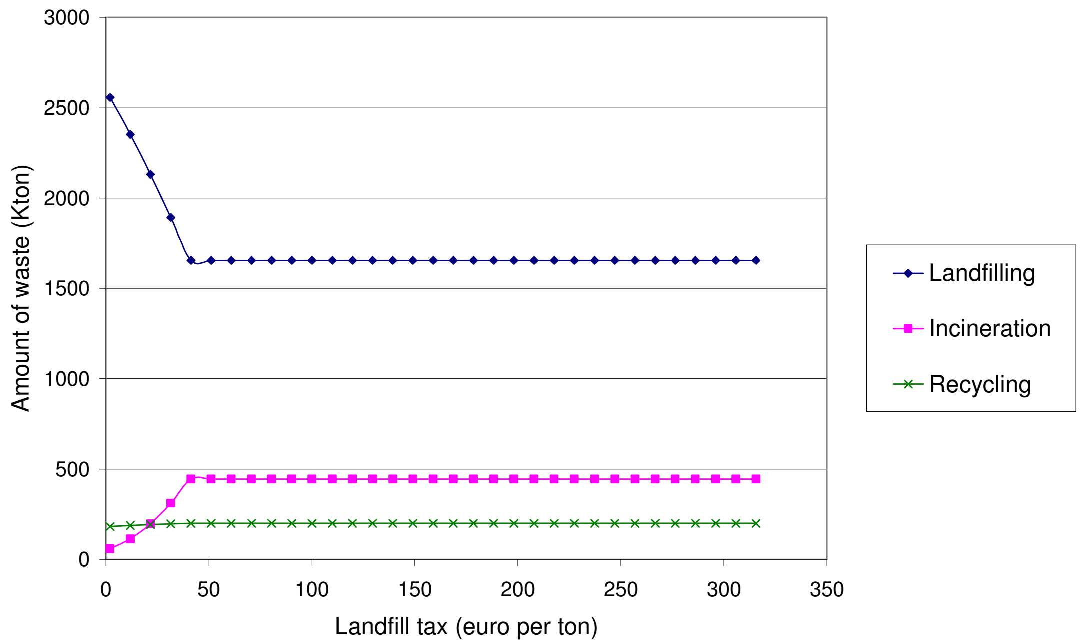


*Figure 4.14 Results scenario 4: impact of waste export on recycling of waste from the service sector.*

Economic theory favours the use of 'positive' policies, like taxation instead of 'negative' ones, like bans. By internalizing the external costs of landfilling in the price of landfilling through taxation, actors in the economy can make an informed choice whether they want to recycle, landfill, or incinerate their waste. A landfill ban, however, forces actors to recycle or incinerate all their waste regardless of whether this situation is optimal (in terms of least costs) or not. This suggests that a ban on landfilling can only be optimal if the government is sure that the optimal level of landfilling is equal to zero and recycling and incineration capacity is large enough to treat all the waste that would usually be landfilled.

In this Chapter we will discuss the effectiveness of the landfill ban as compared to the landfill tax. By introducing a landfill ban in the ex-ante model, we can compare the social costs of a landfill ban as opposed to the social costs of a landfill tax.

Within a general equilibrium framework, the introduction of a landfill ban is similar to setting a landfill tax that is so high that the total demand for landfilling declines to zero. We use the same model as described in Section 4.2. To analyze whether a tax or a ban is more cost-effective, we need to assume that the incineration capacity is large enough to incinerate all waste. If we would not make this assumption, the ban could not be effective because some combustible waste would need to be landfilled due to capacity constraints. Figure 4.16 shows how large the production loss will be if sectors have to recycle and / or incinerate all their waste.

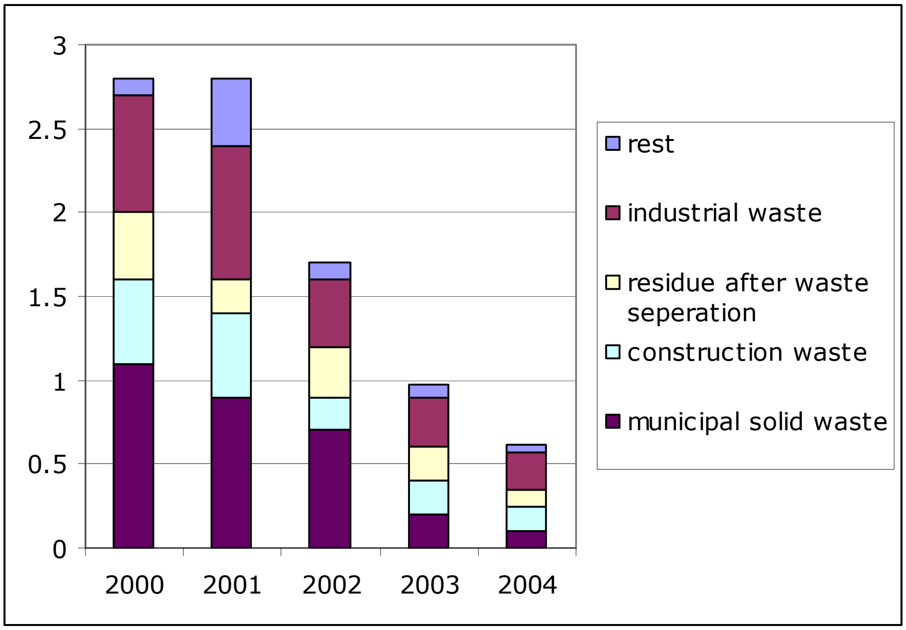


*Figure 4.15 Landfilling of combustible waste in the Netherlands in the period 200-2004. (Source: AOO).*

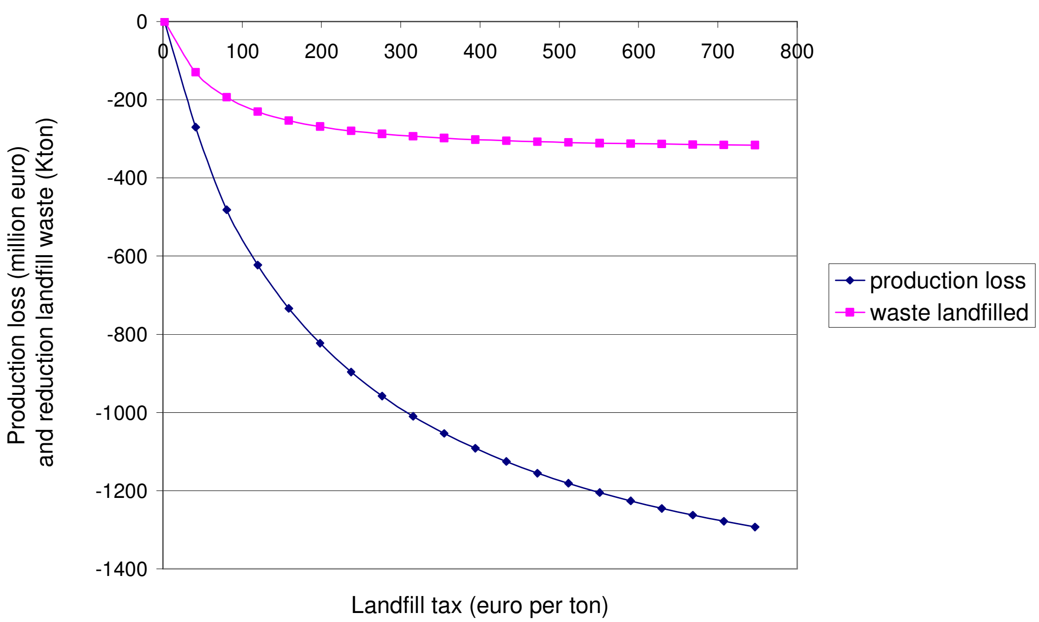


*Figure 4.16 Production loss due to a positive landfill tax or landfill ban.*

If the landfill tax equals about 750 euro per ton, almost none of the waste generated will be landfilled (which is effectively a landfill ban)[33]. To reduce landfilling to zero, the service sector will have to incur considerable costs. The production loss for example is equal to

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[33] With a CES aggregation function it is not possible to reduce landfilling completely to zero. However the amount of landfilling is so low, it can be discounted.

around 1300 million euro in this case. Figure 4.16 shows that reducing the amount of waste landfilled to zero is quite inefficient. Reducing the amount of waste landfilled by about 90% costs about 600 million euro in production loss, while reducing the amount of waste landfilled by 100% costs about 1,300 million euro in production loss.

It is clear that in our model the landfill tax is more effective. A landfill tax of about 80 euro reduces landfilling of waste with about 66% against a relatively slight production loss. Sectors that have relatively high costs of recycling will be able to landfill some of their waste, sectors that can recycle waste relatively cheap will hardly landfill waste.

In Chapter 5 we will also show that the external costs of landfilling are less than the external costs of incineration. Recycling and reuse are also not without costs to the environment. The optimal level of landfilling may therefore not be equal to zero. A landfill tax can be used to reach the optimal amount of landfilling. The landfill ban strictly enforced will not be able to reach this optimal level of landfilling.

## 4.5 Conclusions

The results of the benchmark scenario show that a landfill tax has a significant effect on the amount of waste landfilled. The higher the landfill tax the more waste will be recycled or incinerated. The model predicts that municipalities will start to incinerate all their waste if the landfill tax becomes too high. Only in municipalities that charge a unit-based price for waste collection will the behavior of households be influenced. In these municipalities, households directly notice the effects of the landfill tax by an increase in the price for waste collection and thus will start to recycle more waste. In municipalities that charge a flat fee for waste collection, the model shows that households will not have an incentive to recycle more waste. Recycling efforts, however, are low regardless of the pricing system for waste collection. It is important to remember that the increase in the landfill tax will only provide a small price incentive to recycle. Most of the municipal solid waste is already incinerated so the price increase of waste collection due to the landfill tax will be slight.

The service sectors, in contrast to the municipalities, choose (according to the model calculations) to recycle more waste. Some sectors slightly increase their demand for waste incineration services but the biggest difference in the service sectors is the amount of waste that is recycled. This is mostly caused by the fact that, similar to the Dutch situation, the incineration capacity in the model is too low to accept both an increased amount of municipal solid waste and an increased amount of service waste. If export of combustible waste is allowed, then the service sector will also increase the amount of waste they incinerate. Export of combustible waste will slightly reduce the recycling effort of the service sector.

The model shows that while export of combustible waste will only stimulate producers to incinerate more waste, export of waste to be landfilled has more far reaching effects. If the price of landfilling (including the landfill tax) in the Netherlands exceeds the price of landfilling in the neighbouring countries, the model calculates that all landfill waste will be exported. As a consequence, producers will no longer have a price incentive to recycle or incinerate waste. Thus the landfill tax will no longer be effective.

An increase in the landfill tax will decrease the welfare of our society, measured in terms of consumption. The question remains how large a decrease in welfare is acceptable. The impact analysis shows a relatively large decrease in waste landfilled combined with a relatively low decrease in welfare if the landfill tax is increased from a low level to a slightly higher level. If the landfill tax is higher (for example higher than 100 euro per tonne) the decrease of waste landfilled is much lower compared to the loss of welfare.

A landfill ban can also be used to reduce the amount of waste landfilled. The model shows that compared to the landfill tax, the ban is not nearly as cost-effective. To reduce landfilling to zero may involve higher costs than would be socially optimal. Besides, the optimal level of landfilling may not be equal to zero. A landfill tax can be used to reach the optimal amount of landfilling. The landfill ban strictly enforced will not be able to reach this optimal level of landfilling.

Returning to the research questions that had to be answered by the *ex-ante* analysis, we can conclude as follows:

- What are the conditions that the waste market has to fulfill in order to apply the landfill tax as an effective instrument?

The analysis suggests a number of important conditions. There should be no restrictions on the availability of incineration capacity (both within The Netherlands and abroad). The limited capacity of incineration plants poses a barrier to the effectiveness of the landfill tax because it limits the options producers have to get rid of their waste. Producers cannot increase incineration and therefore only have the option of recycling more waste (which may be a more expensive option). Allowing export of combustible waste and thus effectively solving the capacity problems increases the effectiveness of the landfill tax, assuming that there is sufficient capacity available in neighbouring countries. Flat fee pricing of waste collection also influences the effectiveness of the landfill tax. Households will not increase their recycling efforts if they do not 'feel' the higher cost of landfilling. However, even in municipalities that ask a unit-based price for waste collection the increase in recycling effort is slight. Therefore in this case one can question whether it is necessary to introduce unit-based pricing as the costs of introducing such as system may be higher than the benefits provided by higher recycling efforts. Finally, allowing waste export to be landfilled abroad will restrict the effectiveness of the landfill tax if this effectiveness is measured in terms of 'the reduction in the total amount of waste landfilled'. However, if effectiveness is measured in terms of 'the reduction of the amount of waste landfilled ***in the Netherlands***', allowing waste export for landfilling in other countries would turn the landfill tax into a very effective instrument.

- What is the desired or optimal rate of the waste tax in order to make the suppliers of waste choose alternative waste treatment options (incineration and recycling)? In particular, should the rate be differentiated according to certain aspects such as type of waste, treatment option and waste supplier?

At the present landfill tax rate of more than € 80 per tonne it is already attractive for a lot of waste suppliers to turn to alternatives. However, due to other restrictions (such as a lack of incineration capacity or a ban on the export of waste) they may be forced to landfill their waste anyway. In the absence of such restrictions, higher levels of the landfill tax rate would lead to less landfilling, but at the expense of relatively high social costs. The analysis in this chapter does not allow us to draw conclusions on the impact of a differentiation of the tax rate by type of waste, treatment option or waste supplier. However, economic theory tells us that any differentiation should only be based upon differences in external costs of the waste and the treatment option (see Chapter 5), and not on the type of waste supplier.

- What are the financial consequences (in the short and the long term) of landfill taxation, landfill bans and legal obligations (for landfills and incineration plants) to accept waste, both for the waste suppliers and the waste treatment companies?

In economic terms, a landfill ban is similar to a prohibitively high landfill tax rate (about € 750 per tonne). Our analysis shows that the social costs of reducing the amount of landfilled waste to zero would be very high. Reducing the amount of landfilled waste (from the service sector) by about 90% costs about 600 million euro in production loss, while reducing it by 100% costs about 1,300 million euro in production loss. A similar calculation could not be made for the case of legal obligations to accept waste, but it is obvious that such an obligation would also imply welfare losses (assuming that it would force waste treatment companies to undertake unprofitable operations).

- Which instrument will (in the short and the long term) lead to the lowest costs of waste treatment for the waste suppliers?

A well-designed system of environmental taxes[34] will minimize the total social costs of waste treatment, provided that there are no market distortions. This means that waste suppliers should have the opportunity to choose between alternative treatment options (domestically or abroad) and that the tax rates reflect the external costs of the treatment option (landfilling as well as incineration). The latter issue is the subject of the next chapter.

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[34] Alternatively, tradable landfill permits could be considered. This instrument, which is already applied in the UK, combines the efficiency advantages of a tax with the certainty of a cap on the total amount of landfilled waste.

# 5. External costs of landfilling and incineration in the Netherlands

## 5.1 Introduction

In economics, environmental and social effects are generally defined as external effects. An external effect, or externality, is said to exist if an economic agent's decision has an influence on another agent's well-being or production possibilities and the former does not (properly) take these effects into account. The classic example of an external effect is that of an upstream factory polluting a river that has a negative impact on catches in a downstream fishery. A (negative) externality exists if, in deciding upon how it will produce and consequently how much pollutant it will emit to the river, the upstream factory does not take this effect into account.

Because of its unwanted nature, solid waste is often considered an externality. The extent to which solid waste actually is an externality depends, however, on the method by which it is processed. Clearly, if waste is littered or illegally dumped, the externality will be substantially larger than if the waste is recycled or re-used in a sustainable manner. Policy makers generally use the level of externalities to determine the preferred ranking of waste management options. In the Netherlands, landfilling is considered an environmentally less-favourable option than incineration. Whether the level of externalities of landfilling exceeds those of incineration in the Netherlands is unclear, however.

The present chapter addresses the following research questions (questions 2a through 2c as formulated in Chapter 1):

- What are the social costs and benefits of the following treatment options for household waste and comparable waste from firms:
  - Landfilling all waste;
  - The present situation;
  - Terminating the landfilling of combustible waste by shifting it to waste incineration;
  - Terminating the landfilling of combustible waste by shifting it to co-incineration (in power plants);
  - Terminating the landfilling of combustible waste by optimizing/maximizing recycling.
- To what extent can the current level of the Dutch landfill tax rate be regarded as an internalisation of the environmental costs of landfilling?
- From a social cost-benefit perspective, what is the optimum way of waste treatment for household waste and comparable waste from firms?

The main objective of this component of the study is therefore to determine the level of externalities of waste incineration (including co-incineration) and landfilling in the Netherlands. This objective serves two purposes. First, it allows Dutch policy makers to test the premise that landfilling is less desirable than incineration, as is assumed by the waste hierarchy. Second, because the externalities are expressed in monetary terms, the analysis allows for the comparison of the the total social costs of landfilling and incineration, being the sums of private and external costs. Such a comparison facilitates Dutch policy makers in designing waste management policies, taking into account economic, environmental, as well as social effects.

Externalities are converted into economic effects using the method of 'economic valuation'. In the context of waste management, the foundations of this method are strongly based on the impact pathway approach (COWI, 2000). This approach proceeds sequentially through the lifecycle or pathway of an economic process, linking impacts to burdens, and subsequently valuing these impacts economically.

It should be emphasised that the results are associated with considerable uncertainty. Therefore, the outcome should be treated with great caution. These uncertainties result from necessary assumptions that underlie many of the calculations. Also, certain effects have been excluded from the analysis, due to the lack of data. We do confirm, however, that the most important effects have been included in the study.

This chapter is structured as follows. In Section 5.2 , the main external effects, as well as the methods available to value these external effects are explained. In Section 5.3 and 5.4, the actual external effects for landfilling and incineration are derived. Section 5.5 compares the external costs estimates of this study with foreign estimates and also compares the total social costs of landfilling and incineration. Section 5.6 addresses co-incineration. Conclusions are presented in Section 5.7.

## 5.2 Economic valuation

### 5.2.1 Integrated approach

In this study, economic valuation is adopted as the method to quantify the waste-related externalities. The main reason to express external effects in monetary values is that it allows for the comparison between private costs of various waste management options and the environmental and social costs and benefits related to these options. In this approach, the change in well-being is the basis for valuing external effects. Welfare is expressed in terms of social costs. Social costs are those costs borne by all households and firms in an economy, that is, the private (internal) costs *and* the external costs of an activity. If an externality is present, the marginal private costs and the marginal social costs of an economic activity do not coincide. This results in an inefficient allocation of resources. A condition for an efficient (optimal) allocation of resources in an economy is that the marginal social costs of an activity equal its marginal social benefits.

As mentioned, economic valuation assigns a monetary value to a good or service. One monetary measure of economic value is the Willingness-to-Pay (WTP), which is defined as the maximum amount of money a person is willing to pay to obtain a good or service. An individual's WTP for a good is a reflection of his preferences for this good relative to other goods. An alternative measure of economic value is the Willingness-to-Accept (WTA). WTA is defined as the minimum amount of money an individual requires as compensation in order to forego a good or service. In practice, researchers have encountered serious difficulties in estimating the WTA for the loss of some environmental good (Bateman and Turner 1992). Therefore, WTP is most common in valuation studies.

Ideally, economic valuation forms an integral part of the overall environmental assessment. An example of such a step-wise approach is the ‘impact pathway approach’ (COWI, 2000). Figure 5.1 illustrates the overall procedure of the impact pathway approach. First, overall emission levels and other external effects are determined in physical terms. Then, the impacts of these effects on economic activities and human well-being are assessed. Next, these impacts are translated into monetary values.

An advantage of this approach is that it enables the comparison of the benefits of some environmental improvement with the costs to realise such an improvement. Since many studies have applied valuation methods, standard values can be derived for most pollutants or impacts. These allow translating emissions directly into costs bypassing the elaborate impact pathway approach. A disadvantage of this approach is that the complexity of economic valuation often leads to high degrees of uncertainty of the final results. Moreover, attaching monetary values to, for example for human health, is a politically sensitive issue.

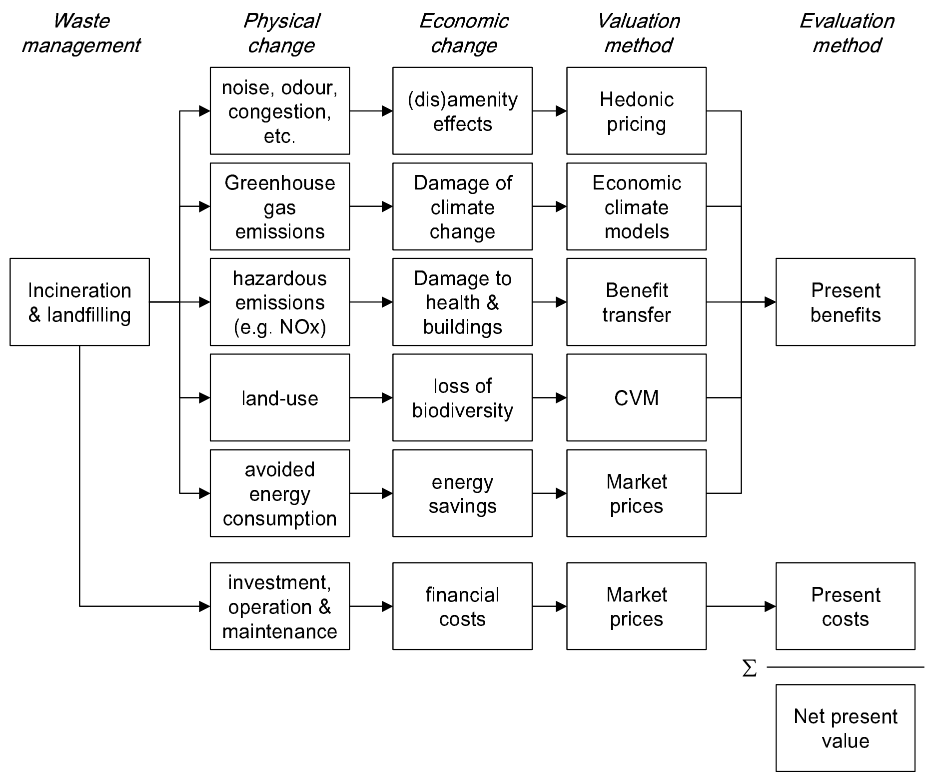


*Figure 5.1 Economic valuation and the impact pathway approach.*

The valuation of external effects encounters various problems. First, because external effects, by definition, occur outside the market, market values for these effects are generally absent. Therefore, special techniques are required for the estimation of external values. Second, external effects of recycling-related processes occur at various locations. It is impossible to value these effects at each individual location. Therefore, estimated values need to be transferred to other locations. In the following, valuation and transfer techniques are explained in more detail.

## Valuation techniques

For the valuation of external effects, various techniques exist. Some values can be directly based on market values. For example, if emissions of air pollutants from a coal fired electricity plant adversely affect agricultural yields in the region, the values of foregone crop losses can serve as a measure of the environmental damage of the electricity plant. Other values can be indirectly valued on the basis of market prices for surrogate products or services. For example, deforestation may lead to a shortage of fuel wood. The alternative fuels, which may need to be imported, may again represent the external environmental costs of deforestation.

In both examples, market prices are directly or indirectly indicative for the external effect. For most external values, however, market values do not exist. The techniques for the valuation of these non-market effects are generally classified into methods that are derived from 'stated preferences' and values that are based on 'revealed preferences' (Freeman 1993). Revealed preference methods calculate external costs and benefits indirectly by using the relationships between environmental goods and expenditures on market goods. This category includes, for example, the hedonic pricing method (HPM), and the averting behaviour method (ABM). Stated preference methods ask the individuals their WTP for the environmental good directly by using structured questionnaires. The contingent valuation method (CVM) is the most important technique belonging to this category. Because of their relevance for waste related externalities, HPM, ABM and CVM are explained in more detail.

The idea underlying the HPM is that the price of a good is a function of its attributes, including environmental attributes (Palmquist 1991). The HPM has been used to analyse house prices. House prices are seen as a function of characteristics of the house itself (e.g. number of rooms, heating system), neighbourhood characteristics (e.g. proximity to schools and shops), and also environmental variables such as ambient air quality, or its proximity to a forest. The HPM proceeds by estimating a so-called hedonic price function by regressing house-price on the relevant characteristics. In the simplest form of the method, a measure of the value of an environmental characteristic of interest can be deduced by differentiating the hedonic price function with respect to the characteristics of interest. If, for example, a landfill causes odour problems in the surrounding neighbourhood, the value of this external cost can be measured by the HPM by statistically isolating the price differential that is due to the nuisance caused by the odour.

ABM values environmental quality by looking at the expenditure people make for goods that can substitute for a decrease in environmental quality. For example, expenditures on bottled water can give an indication of the WTP of people for preventing the adverse health effects from using polluted water. There are numerous problems with this method and in most cases the expenditures on substitute goods will underestimate the true value of a decrease in environmental quality.

The CVM estimates the WTP for a change in the quantity and/or quality of an environmental good by using survey techniques (Mitchell and Carson 1989, Hoevenagel 1994). In a questionnaire a hypothetical change is described and the respondent is asked directly for his or her WTP for this change. Valuation questions are usually supplemented by questions on socio-economic characteristics and relevant attitudes and preferences regarding the good in question. This information is used to estimate a valuation function which 'explains' WTP as a function of these variables. The valuation function can be used for validity checks (for example, testing whether WTP is positively related with income, as theory would predict) and for correcting average WTP in the case of certain response biases (for example, an overrepresentation of high income groups).

In this study, external values generated through various techniques are used. For material and crop damage market prices and ABM values are applied. HPM is used to determine the disamenity value of incineration and landfilling. Values related to human health are mainly derived through CVM. Costs and benefits of global warming, which are generally determined through extensive climate change impact models, use a mixture of the above methods.

### Benefit transfer

Because it is practically impossible to estimate each exposure-response relationship or value at the respective time and place of a particular externality, using data from previous studies focusing on a different region or time period is inevitable. Therefore it is important to know when data from other studies can be used and under what conditions. For the transfer of monetary values, this practice is known as 'benefit transfer' (Navrud 1994).

Two approaches to benefit transfer can be taken (Bergland *et al.* 1995). The first is transferring corrected mean unit values. For example, suppose that a study undertaken in the USA has estimated that the mean unit valuation of an extra asthma attack from increased concentrations of ground-level ozone is US$500. In the absence of a European study, such a unit value could be used to value asthma attacks in a European context by using the exchange rate between US$ and Euro's. If in this example, however, it has been found that there is a relationship between income and the value placed on an asthma attack, the unit value can be corrected for differences in income between the USA and the European country in question. A second approach to benefit transfer is to transfer the complete valuation function determined through meta-analysis. Besides income, most studies find that other socio-economic and demographic factors have an influence on the valuation. Using statistical techniques (regression analysis) some valuation studies estimate a valuation function, that explains the value placed on a good or service as a function of these factors. This function can then be transferred to another site and by inserting the local values of the explanatory variables one can calculate an adjusted value.

The most important variable used in benefit transfer of external values is the income elasticity of demand for the environmental good. Equation (5.1) shows how the adjusted benefit estimate $B_p$ can be calculated for the country $p$ under investigation.

$$B_p = B_s \cdot \left(\frac{Y_p}{Y_s}\right)^{\varepsilon} \tag{5.1}$$

where $B_p$ is the original benefit estimate from the original country $s$, and $Y_s$ and $Y_p$ are the income levels of country $s$ and $p$, respectively, and $\varepsilon$ is the income elasticity of demand for the environmental good in question. For example, a high $\varepsilon$ implies that the external value measured in a developed country will be discounted considerably if the value is transferred to a developing country.

Studies report rather different levels of income elasticities for environmental goods. ADB (1997) assumes the income elasticity to be one. Krupnick *et al*. (1996) reject the notion that there is a proportional variation between the demand for environmental quality and income. Kriström and Riera (1996) find an income elasticity of environmental improvements of less than one in Europe. Pommerehne (1988) suggests an income elasticity of 0.3. Flores and Carson (1997) conclude that an environmental good that is a luxury demand may have an income elasticity of WTP that is greater than one, less than one, or perhaps even negative. Given the indecisive empirical evidence, we use a central estimate of the income elasticity of one, and alternative estimates of two and five. The exchange rates between different currencies will be based on purchasing power parities.

### External effects of waste management

Table 5.1 demonstrates the main external effects of waste management processes. Most waste management processes generate emissions of pollutants to the atmosphere, and sometimes to surface and ground water as well. Moreover, landfill sites and incineration plants often cause significant nuisance or disamenity effects in the vicinity of the locations (Smith *et al.* 1986, Brisson and Pearce 1995). To assess the magnitude and significance of environmental effects, a range of criteria are relevant such as location and timing of the effect, and whether the effect is reversible or irreversible. In this study the main focus is on the actual impact of the externality on societies' well-being.

In the following Chapters various categories of external effects are discussed. An overview of available external values is provided and a selection of the most suitable external values is made for application in this study.

*Table 5.1 Translation of environmental effects to the main effects on well being.*

| Environmental issues | Residual effect on well being |
|---|---|
| Air pollution | Chronic and acute morbidity |
| | Chronic and acute mortality |
| | Occupational health |
| | Damage to buildings, monuments and materials |
| | Damage to forest resources and agriculture |
| Global warming | Mortality and morbidity |
| | Damage to buildings and materials |
| | Damage to forest resources and agriculture |
| Ground and surface water pollution | Safety and availability of drinking water |
| | Recreational value |
| | Biodiversity |
| | Loss of fishery |
| (Dis)amenity | Odour and visual pollution |
| | Noise |
| | Congestion |
| | Willingness-to-recycle |
| | Convenience |

### 5.2.2 Health

Pollution from waste-related activities can have many impacts on human health, ranging from short periods of coughing to premature death. The valuation of human health impacts remains one of the most controversial aspects of any valuation study. Many reactions to the monetary valuation of these impacts are partly caused by the unfortunate choice of terminology such as the 'Value of a Statistical Life' (VSL).

Two main types of studies can be distinguished. First, values for mortality impacts are mostly based on studies using the CVM and the hedonic wage method (HWM). Using CVM, people are directly asked for their WTP to reduce mortality risks. Using the hedonic wage method, differences in wages between professions with high and low mortality risks are used to derive a value for these mortality risks. Note that these studies thus do not value *life* but small differences in mortality *risks*. What they assess, in effect, is the amount of money an individual or household would like to pay, as a sort of insurance premium, to avoid or reduce a small change in health risk.

Second, values for health are derived from the costs that are incurred in treating these impacts, the so-called cost-of-illness (COI) approach. In theory, WTP estimates are to be preferred to COI estimates for the valuation of externalities and cost-benefit analysis. However, for many impacts WTP estimates are not available and one has to resort to COI estimates. For those impacts where both WTP and COI estimates have been made, COI estimates are consistently smaller than WTP estimates.

An important parameter in the monetary valuation is the VSL, which reflects people's WTP for small reductions in mortality risks for themselves and others. A review of studies from Europe and the US, covering three valuation methods (CVM, HWM and consumer market surveys) determines the average VSL at € 3.1 million based on 1997 prices (EC 1995). Two fundamental objections are raised against the VSL method. First, many people whose deaths are linked to air pollution are suspected of having only a short life expectancy even in the absence of air pollution. The VSL ascribes the same value to someone with a day to live as someone with tens of years of remaining life expectancy. Second, the VSL method ascribes the full VSL to air pollution. Air pollution, however, is only one factor of perhaps several that determine the time of death. In this study VSL estimates are only used to value fatal accidents, mortality impacts in climate change modelling, and similar cases where the impact is sudden and where the affected population is similar to the general population for which the VSL applies.

Alternatively, mortality and morbidity can be valued on the basis of 'years of life lost' (YOLL). For quantification of this value it is necessary to interpret the estimate of the VSL as the present value of a number of life years. The YOLL approach is particularly recommended for deaths arising from illnesses linked to exposure to air pollution. The value will depend on a number of factors, such as how long it takes for the exposure to result in the illness and how long a survival period the individual has after contracting the disease. The YOLL value of mortality reduces with age, but not on a proportional scale. In this study, the YOLL approach is used in cases where the hazard has a significant latency period before impact, or where the probability of survival after exposure is altered over a prolonged period.

The health impacts can be broadly subdivided into three categories. First, premature deaths from air pollution are estimated. Second, changes in morbidity result from air pollutants. These effects include respiratory impacts such as coughing, hospital admissions for respiratory infections and asthma attacks, and other health impacts. Third, accidents may occur as a result of related economic activities. Within these categories, the dose-response functions further distinguish between acute and chronic impacts. Acute impacts are those impacts that occur during or immediately after (within a few days) a period of increased pollution levels. Chronic impacts are those impacts that are the result of prolonged exposure to elevated levels of air pollution. Another distinction is made between health impact occurring at the workplace (i.e., occupational health) and health impacts that result for society in general.

### Human health values related to emissions

Toxic emissions are a common cause of damages to human health. Values are derived for the classical 'macro-pollutants': sulphur dioxide ($SO_2$), oxides of nitrogen ($NO_x$: nitrogen oxide, NO, and nitrogen dioxide, $NO_2$), particulate matter ($PM_{10}$), and ozone ($O_3$)[35]. In addition, values are derived for carcinogenic and non-carcinogenic substances (EC 1995, Dorland *et al.* 1997, Dorland and Jansen 1997).

Valuation of mortality impacts from emissions are divided into the following categories:
- Mortality linked to short term (acute) exposure to air pollution;
- Mortality linked to long term (chronic) exposure to non-carcinogenic air pollutants;
- Mortality from exposure to hazardous materials in the workplace;
- Mortality from cancer.

Valuation of morbidity effects consists of the following categories;
- Morbidity from short term (acute) exposure to air pollution;
- Morbidity from long term (chronic) exposure to air pollution;
- Morbidity from exposure to hazardous materials in the workplace.

The characteristics of emissions and the local conditions of the receiving area play a crucial role in the monetary value of the pollutants. Therefore, a distinction is made between human health impact from emissions from transport and emissions from other processes. The latter is depicted for carcinogenic and non-carcinogenic pollutants in Table 5.2. Recycling-related activities are generally extremely transport-intensive. The local and regional (more than 50 km to almost 3000 km from the source) impacts of transport are reproduced in Table 5.3. These include the indirect impacts resulting from $NO_x$ and $SO_2$ emissions through the formation of, respectively, nitrate and sulphate aerosols. Local impacts are an average of various regions in the Netherlands, on urban and rural roads, during peak and non-peak hours (Dorland and Jansen 1997). Because local emissions occur at a low height the concentrations in the local range are high. Therefore, regional damage is considerably less than the local damage (Friedrich *et al.* 1998).

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35 Ozone is not directly emitted as such but is formed from a number of other emissions. The main precursor emissions are $NO_x$ and volatile organic compounds (VOCs). Because of the difficulties in modelling the resulting ozone concentrations, a simplified methodology was used to value damages from ozone, or rather from emissions of ozone precursors.

*Table 5.2 Summary of values for damages to human health by emissions of production processes and electricity generation in the lifecycle (in 2005 € per ton).*

| Pollutant | damage in € per tonne emitted |
|---|---|
| $SO_2$ direct impacts | 1,132 |
| $SO_2$ indirect impacts through sulphate aerosols | 6,375 |
| $NO_x$ indirect impacts through nitrate aerosols | 5,373 |
| $NO_x$ indirect impacts through ozone formation | - 1,495 |
| $PM_{10}$ (directly emitted) | 26,765 |
| VOC indirect impacts through ozone formation | 829 |
| Arsenic (carcinogenic effects only) | 725 – 14,508 |
| Cadmium (carcinogenic effects only) | 6,525 – 43,503 |
| Chromium (carcinogenic effects only) | 145,021 |
| Nickel (carcinogenic effects only) | 1,451 – 14,505 |
| PAH (benzo-a-pyrene) (carcinogenic effects only) | 1,296,110 |

Source: Dorland *et al.* (2000) updated to price level of 2005 .

*Table 5.3 Damage for local and regional range health impacts for vehicle exhaust emissions in € per ton pollutant emitted (in 2005 € per ton).*

| Pollutant | Local health impact | Regional* health impact | Total health impact |
|---|---|---|---|
| $PM_{10}$ | 328,830 | 92,163 | 420,993 |
| CO | 2 | 1 | 3 |
| $SO_2$ (direct effects) | 2,463 | 294 | 2,757 |
| Benzene (carcinogenic effects only) | 633 | 55 | 688 |
| Butadiene (carcinogenic effects only) | 23,730 | 2,260 | 25,990 |
| PAH (carcinogenic effects only) | 6,217,260 | 595,510 | 6,812,770 |
| DME (carcinogenic effects only) | 2,113 | 203 | 2,317 |
| NOx indirect impacts through nitrate aerosols | | 3,564 | 3,564 |
| $SO_2$ indirect impacts through sulphate aerosols | | 4,476 | 4,476 |

*) Regional is defined as more than 50 km from the source.

Source: Dorland *et al.* (2000) updated to price level of 2005.

### Values for occupational accidents and diseases

Occupational health effects are those health impacts that are inflicted on the worker, working on the economic activities that are under investigation. These effects can be categorised into accidents and diseases. Occupational accidents have a direct physical impact with a clear relation between cause and effects. Occupational diseases generally occur as a delayed, long-term response to continuous exposure caused by the related economic activities.

A particular problem in assessing occupational damages is the extent to which these costs might be internalised, for example, through insurance, compensation payments, higher wage rates, etc. In part, internalisation requires workers to be fully mobile (so that they have a choice of occupation) and fully informed about the risks that they face. Available evidence suggests that internalisation is rarely, if ever, complete. With a lack of data on the extent to which internalisation is achieved, total damages are reported instead. Another difficulty in analysing occupational health effects is that only a limited number of case studies have been conducted in the field of waste management.

The different types of processes that need to be valued in terms of occupational health are recycling, waste collection and landfill management, waste sorting and transportation. Waste collection, landfilling and incineration have a more pronounced negative impact on the health of the workers than other economic activities (Cointreau-Levine 1997). A significant relationship between the level of exposure to solid waste and diseases is reported (SWANA 1996, Poulsen *et al.* 1995b). These estimates, however, cannot be explained in terms of reduced days at work. For waste related accidents applicable data have been found. For example, in Denmark average workforce stays at home for 1.7 days every year as a result of injuries, whereas the waste workers were unable to work for 9.5 days annually as a result of impairment (Poulsen *et al.* 1995a).

The value of lost working days due to waste-related activities consists of the direct 'cost of illness' such as the wage level of the worker, the WTP of the suffering related to the injury, and the expenditures on averting and/or mitigating the effects of illness or injury. Van Beukering *et al.* (1998) estimate the overall marginal value of injuries related to waste collection at € 2,122 per worker per year. Assuming a worker to process 1,000 tonnes of mixed waste per year, the occupational injury damage of one ton of waste is € 2.12. This hypothetical value can be considered an underestimation as it excludes the damage of diseases and mortality.

In estimating transport-related accidents, a distinction is made between fatal accidents and accidents causing injury. Regional variations occur as a result of variations in risk of casualties across countries (Kågeson 1993). By combining information on the distance travelled and the risk per vehicle kilometre, the risk per ton of material transported is calculated (Brisson 1997). Multiplying the external values for human health and the risk of casualties, the mortality and injury costs are determined. Examples for various European countries are presented in Table 5.4 for respectively heavy good vehicles (HGVs) with a capacity of 10 tons and passenger cars.

*Table 5.4 External costs from transport (in € per 1000 km).*

| | Mortality | | Serious injury | |
|---|---|---|---|---|
| | HGVs | Passenger cars | HGVs | Passengers cars |
| Belgium | 6.59 | 1.26 | 0.17 | 0.08 |
| Denmark | 1.26 | 0.31 | 0.01 | 0.01 |
| France | 7.54 | 1.00 | 0.09 | 0.02 |
| Germany | 2.20 | 0.63 | 0.07 | 0.04 |
| Italy | 2.51 | 0.63 | 0.23 | 0.14 |
| Netherlands | 0.94 | 0.63 | 0.03 | 0.02 |
| Portugal | 7.22 | 0.63 | 0.64 | 0.79 |
| Spain | 4.71 | 1.57 | 0.25 | 0.20 |
| UK | 1.26 | 0.31 | 0.05 | 0.03 |

Source: based on Holland *et al.* (1997) and Brisson (1997).

### 5.2.3 Material, crop and forest damage

In this Chapter the damage cost of atmospheric emissions on materials, crops, and natural resources is discussed. Figure 5.2 depicts a simplified cause-effect chain. The methodology that is followed to calculate these damages is analogous to that used to estimate damages to human health. The agricultural impact pathway first calculates the incremental pollutant concentrations through dispersion models. Next, dose-response functions are used to determine the affected crop yield per hectare of the so-called stock-at-risk. The impacts result through direct and indirect atmospheric processes. The same spatial procedure is conducted for forest and materials (i.e. buildings, monuments). Finally, these physical impacts are valued using market prices.

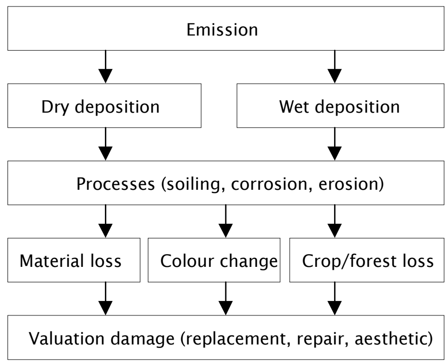


*Figure 5.2 Simplified cause-effect chain for material, crop and forest damage.*

Source: modified from Holland *et al.* 1997.

### Crop damage

The most relevant pollutants causing crop damage are sulphur dioxide ($SO_2$), ozone ($O_3$), and nitrogen oxides ($NO_x$). Hydrogen fluoride (HF) can also cause crop losses, but it is generally thought that these occur only in the direct vicinity of important sources of HF emissions and therefore will be rather small compared to damages from other pollutants.

Deposition of acidic compounds such as $SO_2$ increases soil acidity which in turn influences crop yields. To neutralise soil acidity farmers can apply lime, ground limestone or magnesium limestone. This is a routine agricultural practice that is not practised solely for neutralising deposited acidity from air pollution but also to neutralise soil acidification through agricultural methods. Dorland *et al.* (2000) obtain a damage figure of € 215 per ton of $SO_2$ emitted. This includes damages to the following crops: wheat, potatoes, sugar beets, barley, rye and oats.

It is unlikely that at present concentrations there is a notable direct effect of $NO_x$ on crop yield. $NO_x$ and VOCs, however, do indirectly affect crop yields through their contribution to the formation of ozone ($O_3$). Estimation of reductions in crop losses is carried out at an aggregated level by combining crop-specific dose-response functions for $O_3$ damages with aggregate crop production figures for the European Union. These were derived from experiments in the USA (Dorland *et al.* 2000). A damage figure of € 642 per ton of VOC is obtained.

Deposited nitrogen has a beneficial effect on crop yields because it acts as a fertiliser. The level of this externality is determined by the value of the yield increase due to the deposited nitrogen. Dorland *et al.* (2000) obtain a value of € -697 per ton $NO_x$ (expressed as $NO_2$ mass equivalents). This is not a negative but a positive externality from pollution. It is uncertain whether these fertilisation effects are sustainable in the long term.

### Materials damage

Soiling of buildings, caused by black smoke, is a main damage cost in the category of material damage. The definition of black smoke is based on chemical properties of particles, rather than on their size. The size composition of black smoke can vary considerably depending on the emission source. Roughly speaking, however, black smoke consists mainly of particles with a diameter less than 5 µm. Dorland *et al.* (2000) determine a damage cost of € 662 per ton of particulates emitted in the form of black smoke. This estimate of is based on total UK emissions of black smoke and an assessment of the size of the UK market for cleaning buildings that is completely attributed to soiling from particle pollution (Newby *et al.* 1991).

Dorland *et al*. (1997) calculate the damage costs to materials to be € 260 per ton $SO_2$ emitted from a power plant near Amsterdam. The impact of transport-related emissions on materials is calculated separately at € 702 per kg $SO_2$ emitted (Dorland *et al.,* 2000). This damage cost is based on estimates of the costs of increased repair and maintenance due to incremental concentrations of acidifying pollutants for the following materials: galvanised steel, limestone, mortar, paint, rendering, sandstone and zinc.

## 5.2.4 Disamenity

Disamenity effects of waste related processes are likely to make up a significant share of the externalities caused. Landfill sites or incinerators generate substantial social costs to their neighbouring population. Disamenity effects may occur in different forms. The trucks that transport the waste to and from the sites may cause noise externalities as well as congestion. The landfill site may emit annoying odours and create visual pollution. Increased health risk, or at least an increased perception of higher health risk, is caused for the people living in the vicinity of an incinerator or landfill.

A common way to measure these effects is to use variations in house prices as an indicator of the welfare loss caused by disamenity effects. The risk of following this approach is the potential double counting of external effects. In this study, it is assumed that visual, noise and odour effects are the main cause for the reduction in house prices. These effects are not covered in other categories of external effects.

Brisson and Pearce (1995) provide an overview of several American hedonic price method (HPM) studies on the value of disamenity effects from waste disposal sites. With the exception of one incinerator, the majority of these studies cover landfill sites. Landfilling and incinerating municipal solid waste cause different effects. Households are reluctant to live near an incinerator due to the potential emissions of the highly toxic dioxins. Disamenity of landfill is caused by the perception of ground water pollution and the visual and odour nuisance. Because no valuation data have been found to distinguish between the two waste management practices the overall disamenity value for landfilling and incineration will be assumed equal.

All studies have found a significant effect on house prices owing to the existence of incinerators nearby. House prices increase roughly by 3-4 percent per kilometre distance from a landfill site within a 5.5 kilometre radius. No effect is felt further than 5.5 kilometres away from the site. Regional and site-specific differences, however, are likely to occur. A study on the housing market in a semi-urban area near Milan (Italy) where a large landfilling plant is operated measured effects till 7.5 kilometres from the site (Ascari and Cernuschi 1996). Studies focusing on shorter distances have found very large declines in prices of houses of close proximity to landfills.

Brisson and Pearce (1995) further review three CVM studies of the disamenities of waste disposal facilities. These studies also find that WTP declines with distance to the facility. An important determinant of the WTP is income and the perception of the risk of leachate leaking into water supplies. Households with a high income whose drinking water was at risk are willing to pay substantially more than low-income households dependent on piped city-water. The findings of the CVM studies are consistent with those from the hedonic price studies. Based on the literature survey the following linear regression equation is presented that relates house price depreciation to distance from a waste disposal site (Brisson and Pearce 1995):

$$\Delta \mathrm{HP} = 12.8 - 2.34 \times \mathrm{D} \qquad (5.2)$$

where $\Delta HP$ represents the percentage change in house price and $D$ represents the distance (in kilometres) from the waste disposal facility. The equation suggests a maximum house price depreciation of 12.8 percent at the site of the facility and that there will be no effect on house prices beyond a distance of 5.5 kilometre from the facility.

Based on the disamenity function the annual value of reduction in the real estate prices is calculated. As shown in Figure 5.3, eight categories of household density are combined with four levels of house prices (CBS 1996). The overall values are converted to annual values by taking 8 percent of the total reduction (Jansen 1988). The variation is substantial. For a landfill which is located in a neighbourhood with a density of 125 houses per $km^2$ and an original house price of € 50,000, the reduction in real estate value is € 1.8 million. For the other extreme, a density of 1000 houses per $km^2$, having an average value of € 200,000, the reduction is € 59 million.

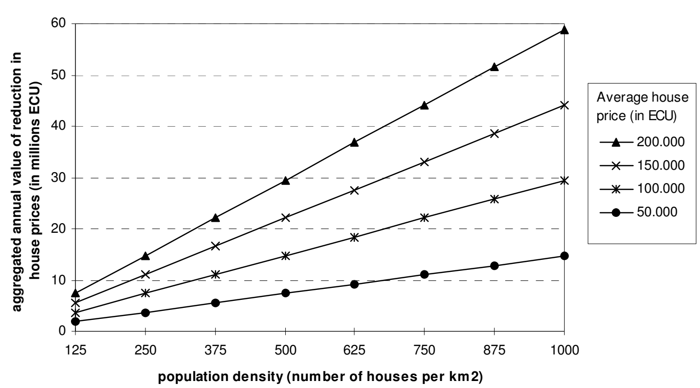


*Figure 5.3 Annual disamenity value of landfill sites.*

To enable the comparison between incineration and landfilling, external costs are calculated on a per unit basis. This step in the analysis is rather uncommon for the disamenity effect. In reality the disamenity is not primarily determined by the amount of waste processed by the landfill or the incinerator, but by the sheer existence of the site. To facilitate comparison, however, the disamenity value is assumed to be proportional to the total amount of waste processed. Values reported in the literature vary significantly. A study on landfilling in Minnesota calculated a range between € 1.2 and € 3.1 per ton of landfilled waste (IIED 1996). The study near Milan (Italy) estimated external costs to be € 10.6 per ton of solid waste (Ascari and Cernuschi 1996).

Despite these earlier studies, a study into the external effects of landfills and incinerators for the European Commission in the year 2000 (COWI, 2000) reported a 'huge deficiency' of European studies to measure the disamenity costs of landfills and incineration plants. The 'huge deficiency' of European studies was partly addressed in 2003 by a large UK study on the disamenity costs of landfills for DEFRA (Cambrdige Econometrics *et al.*, 2003). This study employed the HPM method for 6,100 operational landfills using data on 592,000 housing transactions in the 1991-2000 period. It estimated disamenity costs of between € 2 and € 3 per tonne of landfilled waste (in 2003 prices). Apart from this central result, the study also presented a number of other interesting results:

- Large regional differences in disamenity costs were found. Particularly, the distance-effect on house prices (see Eq. 5.2) is not stable and depends very much on the characteristics of the property market under consideration;
- No statistically significant differences in disamenity costs between hazardous and non-hazardous landfill sites could be found;
- Disamenity costs tend to be highest around the opening of new sites and level off after some time when local residents adjust to the presence of the landfill. After closure of the site, disamenity costs tend to vanish rather quickly, as long as there are no long-term problems such as water contamination or gas seepage and the area is appropriately landscaped;
- Disamenity effects were measured within a radius of 0.8 km (half a mile) from the site. Beyond that distance, no effects could be measured. This distance corresponds relatively well with the suggestion from a recent WHO report that any potential exposure from landfill sites is limited to 1 km from the site by the air pathway (WHO, 2001).

The calculations of disamenity costs per unit of waste processed though landfilling and incineration are described in Section 5.3 and 5.4, respectively.

### 5.2.5 Transport-related externalities

Transport is a major cause of negative externalities related to waste management. For example, as shown in Table 5.5, incineration involves the transport of various types of materials and products over long distances. The majority of the externalities are caused by road transport. Externalities of alternative transport modes such as ships and railway are negligible. The transport of incineration associated waste involves 15% trains, 25% water, and 60% road transport.[36] Assuming that the waste is transported with a standard diesel 10 ton truck, and taking into account the transport of the residues, such as fly ash and bottom ash, an average ton of incinerated waste involves 20 kilometres of road transport (i.e. 60% of 34 kilometres). Because landfilling does not involve the transport of major inputs or waste residues, the transport intensity is substantially lower. The proc-

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36 Personal communication E. Zoontjes, Vereniging Afvalbedrijven, August 2005.

essing of one ton of landfilled waste involves road transport over a distance of 15 kilometres (i.e. 60% of 25 kilometres).

*Table 5.5 Transport characteristics (tonne / year).*

| Transported material | Distance | Quantity |
|---|---|---|
| Municipal waste transport | 15 km | 465,000 |
| Lime transport | 400 km | 1,888 |
| Fly ash | 200 km | 9,066 |
| Filter residue + salt residue | 50 km | 5,500 |
| Furnace bottom ash | 50 km | 120,267 |

Source: Van Beukering *et al.* 1998.

The externalities related to waste transport include four categories:

- Climate change caused by the emission of $CO_2$;
- Health and material damage caused by various air emissions;
- Mortality and morbidity caused by accidents;
- Congestion.

Because the health and material damage externalities have been discussed in the previous Chapters and climate change externalities will be discussed in Section 5.2.6, only congestion will be explained here in more detail.

### Congestion

Congestion damages occur from three stages of the incineration life cycle: the construction activities, the daily waste transport activities, and the disposal of residues. The external costs of congestion result from various effects. The most important costs are the time costs of delay. Indirect effects include increased emissions levels and danger in traffic (Daniel and Bekka 1998). Table 5.6 summarises two studies that provide measures for the marginal costs of traffic congestion (Greene *et al.* 1997). Newbery (1995) estimates the marginal congestion cost for 'other rural roads' and 'urban central' roads at peak times at € 7 and € 5371 per 100 passenger car unit kilometre (PCUkm), respectively. Newbery suggests that one heavy good vehicle kilometre (HGVkm) is equivalent to two PCUkm. Brossier (1996) provides for intercity traffic estimates of marginal congestion costs of trucks averaged over a year on various types of roads and the results ranges from € 2.95 and € 17.1 per 100 HGV.

*Table 5.6 Marginal costs of congestion in the UK, 1990.*

| | Marginal Congestion Costs (€/100 HGVkm) |
|---|---|
| Newbery (1995) | |
| - rural road | 7.2 |
| - urban central peak | 5373.0 |
| Brossier (1996) | |
| - rural road | 5.85 |
| - motorway | 2.95 |
| - national road | 17.1 |

Source: Greene *et al.* 1997, p.83.

It is assumed that all transport of solid waste takes place in a ‘non-central’ area with a truck of 15 tonnes during ‘off-peak’ traffic hours. We applied Newbery’s estimates (1992) of 0.29 and 0.53 euro per HGVkm for respectively ‘non-central off-peak’ and ‘central off-peak’ traffic. Given the difference in population densities, we apply the non-central value for the Western European country and the central value for the Southern European country. We assume, as Newbery (1990) suggested, that one HGVkm equals two PCUkm. The annualised values are summarised in Table 5.7. Distances for the various activities are considered to be equal. Adopting an annual quantity of 5 million tons of waste, the congestion costs of incineration in the Netherlands amount to € 0.23 per ton of processed waste.

Because quantities incinerated and landfilled in the Netherlands are similar, Table 5.7 also provides sufficient basis for the estimation of the congestion costs for landfilling. Recognising the fact that the disposal of ashes is not relevant for landfilling, the total congestion costs for landfilling amount to € 0.13 per ton of waste.

*Table 5.7 Annual congestion costs for incineration in the Netherlands.*

| Congestion costs | Annual congestion costs (*€ per year*) | Congestion costs (*€ per ton of waste*) |
|---|---|---|
| During construction of incinerator | 260,892 | 0.01 [a] |
| During operation of incinerator | 601,691 | 0.12 |
| During disposal of residues (i.e. ash) | 489181.5 | 0.10 |
| Total | 1,351,764 | 0.23 |

[a] The congestion costs caused during the construction phase are spread equally across the waste incinerated during the period of 10 years.

Source: based on Newbery (1995).

Note that Newbery’s estimates are valid for road conditions in the United Kingdom. As no similar data for other countries in Europe were found, UK data were adopted for the Netherlands. One potential modification could be made for the value of delay caused by congestion, based on difference in wage levels between European countries. However, in this particular case study we assumed, in line with the principles adhered to in the ‘ExternE’ project (Dorland *et al.,* 2000), one uniform value for the whole of Europe, without correcting for wage levels.

### Transport externalities summarised

On the basis of the above estimates, physical and monetary standard values have been derived to determine the overall value of transport related externalities (see Table 5.8). The main contributors to transport-related externalities are health effects (i.e. $NO_x$, $PM_{10}$) and congestion. Mortality and morbidity caused by road accidents are negligible. The total externality caused by the transport of one ton of waste over a distance of 1,000 kilometre is almost € 85.

## 5.2.6 Global warming

Global warming is an important impact category in the valuation of externalities related to waste management (Walz *et al.* 1996). Methane ($CH_4$) emissions from landfills contribute approximately 16 percent to the world's total methane emissions (WRI 1996). Depending on the efficiency of the incinerator and the composition of the burned materials, 'waste-to-energy' practices may avoid carbon dioxide ($CO_2$) emissions through supplying electricity to networks.

*Table 5.8 Transport-related externalities.*

| Air emissions | Ton emission/1000km ton | €/ton emission | €/1000km ton waste |
|---|---|---|---|
| $CO_2$ | 0.128000 | 10 | 1.2800 |
| CO | 0.000482 | 0.71 | 0.0003 |
| $SO_2$ transport | 0.000138 | 7,233 | 0.9982 |
| NOx transport | 0.002040 | 3,564 | 7.2706 |
| HC | 0.000322 | 680 | 0.2190 |
| $PM_{10}$ | 0.000146 | 420,993 | 61.4649 |
| Serious injuries | n.a. | n.a. | 0.0300 |
| Mortality | n.a. | n.a. | 0.9400 |
| Congestion | n.a. | n.a. | 11.3750 |
| Transport externalities | | | 83.578 |

Source: Dorland *et al.* 2000, Brossier 1996, Holland *et al*. 1997, Tol, 2005.

Comprehensive climate models that are linked to economic models calculate the costs of climate change. They include agricultural damage, increased morbidity and mortality, damage caused by sea level rise and by extreme events, and loss of species. Various estimates of marginal costs of greenhouse gas emissions have been reported, expressing the average additional cost of a small change in an exogenous scenario, also called the 'business as usual' scenario. These estimates are usually based on the so-called $2\times CO_2$ scenario, that is, a doubling of $CO_2$ concentrations relative to pre-industrial concentrations. Due to the still existing scientific lacunas with regard to global warming, the uncertainties of the results of these estimates are larger than for most other external effects.

In this study, the marginal costs of greenhouse gas emissions from waste deposited at landfills and waste incineration are estimated in the following stepwise approach:

- First, it is examined how much $CO_2$ and $CH_4$ are emitted from one ton of waste processed in Dutch landfills or incinerators;
- Second, the marginal cost ratio between $CO_2$ and $CH_4$ is examined;
- Third, the literature is surveyed for estimates of marginal global warming damages of greenhouse gases per ton of $CO_2$;
- Fourth, the *net present value* of future global warming damages due to the current landfilling and incineration of one ton of waste in an average Dutch landfill or incinerator is estimated.

Note that the estimates in each step are surrounded by large margins of uncertainty. The final estimate of marginal global warming damage per ton of waste is therefore subject to a very large, *compounded* range of uncertainty. In the following subSections, we will provide an indication of the contribution of each step in the calculation of the overall uncertainty of the estimate. Only the quantification of the greenhouse gas emissions of landfilling and incineration are discussed in Sections 5.3 and 5.4, respectively.

### Conversion ratio between $CO_2$ and $CH_4$

The global warming potential (GWP) of $CH_4$ is 21 times higher than that of $CO_2$. Because of different time-damage profiles of $CH_4$ and $CO_2$ emissions [37] and the fact that damages are discounted and GWP is not, this does not automatically imply that the marginal damage per ton of $CH_4$ emissions is 21 times the marginal damage per ton of $CO_2$ emissions (Manne and Richels, 2001). The DEFRA study (Enviros Consulting *et al.*, 2004) reports damage conversion factors in the literature ranging between 11 and 20, and suggests a central conversion factor of 16.5. This means that, in terms of damage, the emission of one ton of $CH_4$ is *equivalent* to the emission of 16.5 tons of $CO_2$. However, Sygna *et al.* (2002) suggest that the ratio of marginal damage costs of $CH_4$ to $CO_2$ *exceeds* that of their GWP ratio: they suggest that the marginal damage cost of one ton of $CH_4$ is 30 times larger than that of one ton of $CO_2$ (Sygna *et al.*, 2002). A recent note of the Dutch government also recommends a conversion factor of 30 (Tweede Kamer, 2004). Most of the literature, however, uses a damage conversion rate of 21 (or close to 21) between $CO_2$ and $CH_4$. We will use this rate for our 'best' estimate.

### Marginal costs of global warming

The global warming costs of greenhouse gas (GHG) emissions include agricultural damage, increased morbidity and mortality, damage caused by sea level rise and by extreme events, and loss of species. Due to the still existing scientific lacunas with regard to the effects of global warming, the uncertainties of the results of these estimates are larger than for most other external effects. In 1995, the Intergovernmental Panel on Climate Change (IPCC)[38] reported marginal damage costs in a range of \$5 to \$125 per ton of carbon (\$/tC).[39] The U.K. government recently issued a formal recommendation on the unit costing of global warming damage from $CO_2$ emissions of ₤ 70/tC (= \$125/tC)[40], based on a literature survey by Clarkson and Deyes (2002). In a recent policy paper, the European Commission quotes marginal damage figures between € 14-20 per ton of $CO_2$ at the lower end to € 80 per ton ('*and very possibly much higher*') at the upper end (European Commission, 2005).

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[37] The average lifetime of $CO_2$ molecules in the atmosphere is over 100 years, while that of $CH_4$ is only 13 years (Manne & Richels, 2001).

[38] The Intergovernmental Panel on Climate Change (IPCC) was established by the United Nations to assess scientific, technical and socio- economic information relevant for the understanding of climate change, its potential impacts and options for adaptation and mitigation.

[39] Prices of 1990.

[40] Prices of 2000.

Tol (2005), on the other hand, concluded on the basis of a survey of 28 studies, that it would be "unlikely that the marginal costs of $CO_2$ emissions exceed \$50/tC and are likely to be substantially smaller than that." [41]

In today's prices, the IPCC range would be around € 1 and € 32 per ton of $CO_2$.[42] The UK recommended unit cost would be € 29 per ton of $CO_2$,[43] the European Commissions' range between € 14 and € 80 per ton of $CO_2$, and Tol would argue that marginal $CO_2$ damages would be less than € 10 per ton. In comparison, the prices at which $CO_2$ emissions permits are currently traded in the European Emissions Trading scheme are around € 19 per ton (www.emissierechten.nl, 23 June 2005).

It is generally assumed that marginal damages of greenhouse gas emissions increase over time. This is because, on the one hand, $CO_2$'s global warming potential increases with the total stock of carbon in the atmosphere, and, on the other hand, the monetary value of global warming damage increases with growing incomes. Simply put, there will be more damage and there will also be more to be damaged. Clarkson and Deyes (2002) suggest an annual increase of one pound on their marginal damage estimate of £ 70/tC.

## Net Present Value

To calculate the *net present value* of the damage costs of greenhouse gas emissions from waste over time, damage costs of future emissions have to be discounted. In doing so, the time-scale is relevant from two perspectives. First, the time frame of the emissions related to a waste management and recycling process varies. For example, in the case of landfilling, the activity takes place today but the actual emission of $CH_4$ occurs in a later stage. Second, the impact of these future emissions is higher than present emissions. This is partly caused by increased radiative forcing of greenhouse gas emissions. Also the ecological and economic background systems become more sensitive to climate changes.

The discount rates used in climate studies vary between 0 to 10 percent. The most commonly used discount rate lies between 3 and 5 percent. The official discount rate for large infrastuctural projects in the Netherlands is 4 percent (Eygenraam *et al.*, 2000). We evaluate discount rates of 3, 4, and 5 percent.

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41 Apart from uncertainty on physical dose-response relationships, the estimates differ because of different assumptions on discount rates and on different assumptions on the marginal utiliy of income across regions. In addition it is uncertain how people value risks, especially concerning risky events with low probabilities and high impacts. The lower (higher) the discount rate, the greater (smaller) the difference in marginal utility of income between rich and poor people, and the more (less) risk-averse people's preferences are, the higher (lower) is the estimate of marginal damage.

42 CPI index USA 1995-2004 = 1.25; exchange rate Euro:US\$ = 0.75115 (March 21, 2005); conversion rate $C:CO_2 = 12/44$.

43 CPI index UK 2001-2004 = 1.04; exchange rate Euro:UK£ = 1.44263 (March 21, 2005); conversion rate $C:CO_2 = 12/44$.

## 5.3 Landfilling

### 5.3.1 Introduction

Landfilling in the Netherlands has declined significantly in the last decades. At present, the Netherlands has 30 active landfill sites which in total process around 4.7 million tons of solid waste per annum. Fifteen years ago, 80 active landfill sites still processed 14 million tons of solid waste. This decline is mainly due to increased recycling and incineration of household waste. Similar to incinerators, landfill sites in the Netherlands are strategically distributed across the country to minimise transport costs (see Figure 5.4).

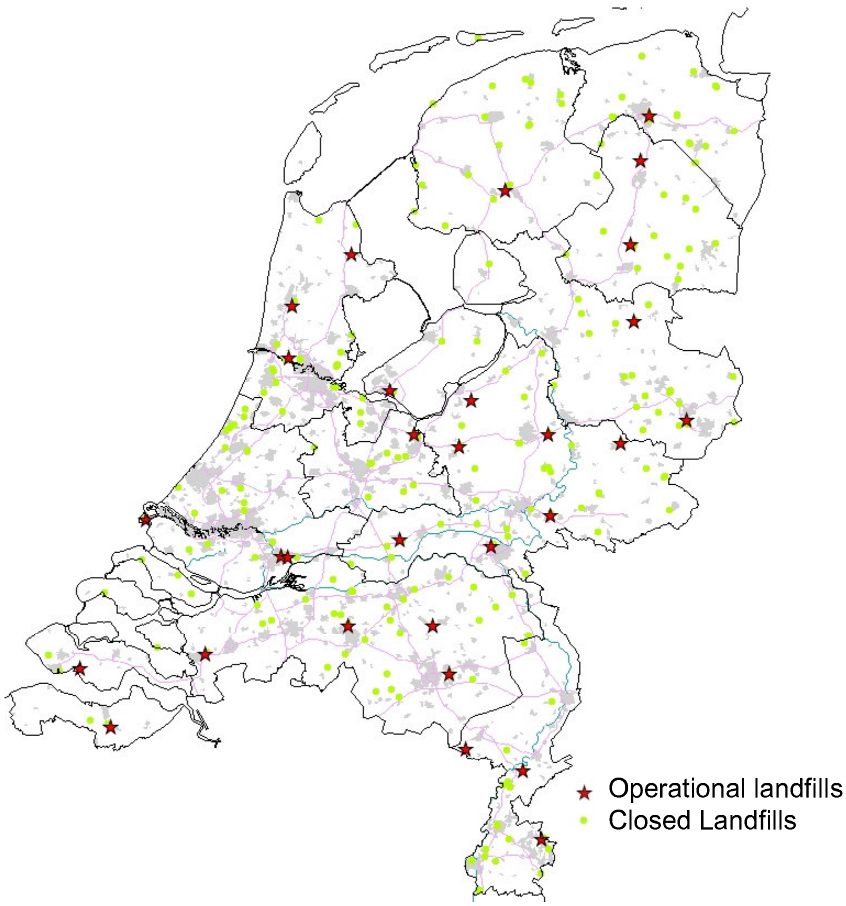


*Figure 5.4 Operational and closed landfills in the Netherlands.*

In the following chapters, the most important environmental effects of landfills in the Netherlands and their economic valuation are addressed. These include:

- Health;
- Disamenity;
- Climate change;
- Land-use;
- Avoided emissions by power generation.

### 5.3.2 Health

The potential impacts of landfills on the health of local residents have been subject of a relatively large number of studies. Recently, De Bont and Larebeke (2002) of Ghent University reviewed the evidence presented in international health research. The reported health effects include: increased risks of birth defects, cancer, respiratory and cardiovascular diseases, and various negative effects on physical well-being, such as nausea and headaches, insomnia, lower respiratory problems, gastro-enteritis, and psychological problems.

While most of these effects have indeed been observed in empirical studies, De Bont and Larebeke (2002) argue that it is in general not easy to determine whether the found symptoms are directly caused by exposure to landfill emissions, whether they are caused by stress and anxiety of living near a landfill, or whether there are other confounding variables (life style, other sources of pollution, and socio-economic factors) that were not properly addressed in the empirical studies. In general, too little is known about the emissions of specific pollutants from landfills and the level of human exposure to these pollutants. Because of the lack of firm dose-effect relationships, it is also difficult to establish the exact area in which people are at risk: different studies use different study areas, from 0.4 to 3 km from the landfill site. A recent WHO study suggests that any potential exposure from landfill sites is limited to 1 km from the site by the air pathway (WHO, 2001).

Despite these shortcomings, De Bont and Van Larebeke, Bont conclude from their literature review that research offers a ‘reasonably certain’ link between the vicinity of a landfill and an increased risk of birth defects. For other health effects, this link would be less clear.

Because of the huge uncertainties regarding health effects, we make an illustrative calculation only, based on the following assumptions:

1. There is no seepage from Dutch landfills to ground or surface water; the only potential pathways are air pollution and stress and anxiety;
2. The only potential health effects are birth defects ( De Bont and Larebeke, 2002), particularly a low birth weight. The increased risk of a low birth weight in the vicinity of a landfill is 30 percent (De Bont and Larebeke, 2002).[44] The ‘willingness-to-pay’ to avoid this particular health effect is valued at € 600,000 per case (Enviros Consulting *et al.*, 2004);[45]
3. The potential impact area is restricted to a distance of 1 km from the site (WHO, 2001);
4. We further *assume* that the fertility and standard (non-landfill related) health status of women living in the vicinity of landfills are identical to the fertility and health status of women in the entire Dutch population.

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[44] This means that the risk of low birth weight in the vicinity of a landfill is 30 percent higher than the risk of a low birth rate in the entire Dutch population.

[45] This is the upper limit of willingness-to-pay for the category ‘all birth defects’, reported in a study for DEFRA (Enviros Consulting *et al.*, 2004).

The average risk in the Netherlands of giving birth to a child with a low birth weight (< 2500g) is 5.7 per cent and the risk of a very low birth rate (< 1500 g) is 0.4 percent. The annual number of births per household in the Netherlands is 0.02999.[46] A GIS analysis carried out for this study showed that a total number of 20,355 houses is located within a radius of 1 km from active Dutch landfills (see Appendix II). Simple calculations show that the expected annual increase in the number of low birth weights due to landfills is 10 cases and the expected annual increase in the number of very low birth weights due to landfills is 0.7 cases.

It stands to reason that the willingness-to-pay to avoid a very low birth rate would exceed the willingness-to-pay for a low birth rate. We do not have enough information to differentiate between the two defects, however. Therefore, we will value both low birth rate and very low birth rate at the same amount of money: € 600,000. The willingness-to-pay to avoid low birth due to landfills rate would then be € 6.2 million and the willingness-to-pay for very low birth rate would be € 0.4 million. Per ton of waste deposited (4.8 million ton in 2003), the willingness-to-pay per ton of waste would then be € 1.30 and € 0.09, respectively.

It is possible, of course, that the assumed willingness-to-pay of € 600,000 to avoid low or very low birth rates is an underestimate of the true willingness-to-pay. But the medical evidence of the relationship between landfills and birth defects (and other health effects) is rather thin, however. It is difficult to determine a ‘best’ estimate: with hesitation we pose the average between the two values found, i.e., € 0.70 per ton of waste.

We also note that some (minor) health effects are probably picked-up by the next external effect that is examined in this Chapter: the so-called disamenity effect.

### 5.3.3 Disamenity

Local residents may experience discomfort from living nearby a landfill. As we saw above, some studies report an increase in the incidence of physical and psychological complaints in the local population. In addition, a landfill may be visually unattractive, and may be a source of noise, dust, odour, and vermin. Together, these various negative effects on well-being are commonly labelled disamenity effects. A number of studies has tried to estimate monetary value of these disamenity costs by measuring the willingness-to-pay of local residents to avoid living near a landfill. Basically, two valuation methods have been employed: CVM and HPM (see Section 5.2).

In Section 5.2, a relatively large and recent UK study into the disamenity effects of landfills has been described (Cambridge Econometrics *et al.*, 2003). Its central estimate of disamenity value was between € 2 and € 3 per ton of landfilled waste.

How can the British disamenity estimate be ‘transferred’ to the Netherlands? There might be various differences between the British and the Dutch situation, both with respect to characteristics of the landfills and to characteristics of the population at risk. With respect to characteristics of the landfills we have little information on the precise

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[46] In the year 2003, 200,297 children were born in a total population of 6,699,686 households (CBS Statonline).

nature of the differences, as well as on the effects of differences on the disamenity costs. As was pointed out above, the British econometric study was even unable to find statistically significant differences in disamenity costs between hazardous and non-hazardous sites.

We do have some information on the characteristics of the population at risk. We do not know whether the basic preferences with respect to living near landfills systematically differ between the British and Dutch populations. We might assume, however, that willingness-to-pay to avoid living near a landfill might be systematically affected by income (i.e., increase with income). Moreover, even if willingness-to-pay *per household* would be perfectly identical between the UK and the Netherlands, then the total willingness-to-pay to avoid living near a landfill would certainly depend on *the number* of households affected.

To implement the benefit transfer function (Eq. .5.1) that was presented in Section 5.2, we used Eurostat statistics on capita gross domestic product (GDP) at purchasing power parities (GDP per capita in PPS) for EU Member States. For the year 2003, the ratio between the Netherlands and the UK was 121/118 (EU25=100), a relatively small difference. [47] Assuming income elasticities $\varepsilon$ of the demand for environmental goods of 1, 2, and 5, respectively, the disamenity costs per ton of landfilled waste in the Netherlands would be 2.5 %, 5.1%, and 13.4% higher than in the UK.

In the UK, the housing density near landfills is approximately 79 houses/km$^2$. [48] In our GIS-based estimate of housing density near landfills in the Netherlands, we found a housing density within a 1 km radius around landfills of 108 houses/km$^2$. Hence, it seems that the Dutch population at risk per ton of waste landfilled is 37 percent (108/79 – 1) higher than the British population at risk.

Combining the effects of income and population at risk (and assuming all other potential effects to be equal), we find that the disamenity costs per ton of landfilled waste in the Netherlands might be 39 percent to 50 percent larger than in the UK. In absolute values this would point to disamenity costs of € 3.5 to € 3.8 per ton of landfilled waste. This estimate is lower than suggested by the (older) COWI study (COWI, 2000), but in line with (and slightly higher than) the estimate suggested by the DEFRA study (Enviros Consulting *et al.*, 2004).

### 5.3.4 Climate change

The volume of greenhouse gas (GHG) emissions from waste is subject to uncertainty. Spakman *et al.* (2003) present the methodology that is used to compute $CH_4$ emissions from landfills for the official climate change emissions registration. This methodology computes current $CH_4$ emissions from all waste deposited in the past (the results are in Table 5.9). For our marginal damage estimate, we are interested in the future emissions of waste currently deposited. This methodology is explained in Appendix I. A major assumption for this latter calculation is the rate of $CH_4$ recovery that is currently applied

[47] Eurostat News Release, 145/2004 – 3 December 2004.

[48] Calculated from data in Cambridge Econometrics *et al.* (2003). This density is very near the average EU density of 80 houses/km$^2$ assumed in the COWI study.

and will be applied in the future on currently active Dutch landfills. Due to technical progress in the Dutch landfilling industry, this rate of recovery is likely to be larger than the historical rate that can be derived from the data in Table 5.9 (about 20 percent in 2002). Dijkgraaf and Vollebergh (2004b) use a $CH_4$ recovery rate of 'best-practice' landfills of 77 percent.[49] Dijkgraaf and Vollebergh's landfill is a 'best-practice' landfill, its technical performance is therefore likely to exceed the average environmental performance of all current Dutch landfills (active and closed). We use estimates of ERM Nederland that suggest an average recovery rate of between 40 and 43 percent (ERM Nederland, 2000).

The share of methane emissions in the total GHG emissions of the Netherlands is 3 percent, and is therefore considered a 'major key source' (RIVM, 2004).[50] Table 5.9 presents data on landfilled waste and $CH_4$ emissions from landfills in the Netherlands from 1990 to 2002. Although $CH_4$ from landfills is still considered a major key source of GHG emissions, Table 5.9 shows that the volume of landfilled waste has decreased over the last decade, while the recovery rate of $CH_4$ emissions from landfills has increased, so that the net emissions of $CH_4$ to the atmosphere have decreased substantially, from 572 Gg in 1990 to 345 Gg in 2002 (–43%).

*Table 5.9 $CH_4$ emissions from landfills in the Netherlands, 1990-2002.*

| Substance | 1990 | 1992 | 1994 | 1996 | 1998 | 2000 | 2002 |
|---|---|---|---|---|---|---|---|
| Waste (Tg) | 14.0 | 11.8 | 9.2 | 6.7 | 5.5 | 4.8 | 4.7 |
| $CH_4$ emissions (Gg) | 572 | 561 | 527 | 483 | 448 | 389 | 345 |
| $CH_4$ emissions recovered/flared (Gg) | 26 | 45 | 64 | 69 | 54 | 66 | 69 |

Source: RIVM, 2004.

Table 5.10 summarizes the assumptions that are used in estimating global warming damages from landfills. The table reports 'low' and 'high' estimates of various assumptions and also a 'best' estimate. The 'best' estimate of global warming damages of landfills is based on the 'best' estimates of the underlying assumptions. This 'best' estimate is € 4.21 per ton of waste. As may be clear from the above discussion, there is still a considerable amount of uncertainty regarding global warming damages that is illustrated by the extremely wide difference between the 'low' estimate (€ 1.46) and the 'high' estimate (€ 54.50 per ton of waste). The difference between the 'low' and 'high' estimates is € 53.04 per ton of waste.

49 Dijkgraaf and Vollebergh assume an annual landfill gas production of 148 $m^3$ per ton waste, of which 78 $m^3$ is exploited, 36 $m^3$ is flared, and 34 $m^3$ is emitted to the atmosphere. Hence, net emissions are 34/148*100 = 23 percent of landfill gas production and the recovery rate is 100 – 23 = 77 percent.

50 A 'major key source' is a technical term used by IPCC to rank emissions of GHGs according to a number of criteria (RIVM, 2004).

*Table 5.10 Assumptions in the estimation of global warming damages of landfill.*

| Assumption | 'Best' | Low | High |
|---|---|---|---|
| Global warming damage €/t$CO_2$ | 10 | 5 | 80 |
| Annual increase in damage | 1 % | 1 % | 1 % |
| $CH_4$/$CO_2$ damage ratio | 21 | 16.5 | 30 |
| $CH_4$ Recovery rate | 42.5 % | 45 % | 40 % |
| Discount rate | 4 % | 5 % | 3 % |
| Global warming damage €/ton waste | 4.21 | 1.46 | 54.50 |

Figure 5.5 presents the distribution of the sources of uncertainty of the estimate of global warming damages per ton of waste over the different assumptions.[51] Figure 5.5 shows that two-thirds of the uncertainty can be attributed to uncertainty over the damage costs of global warming. 23 percent of the uncertainty can be attributed to uncertainty over the $CH_4$/$CO_2$ damage conversion ratio. the remaining uncertainty can be attributed to different assumptions on the rate of discount and the recovery rate of $CH_4$ from landfills.

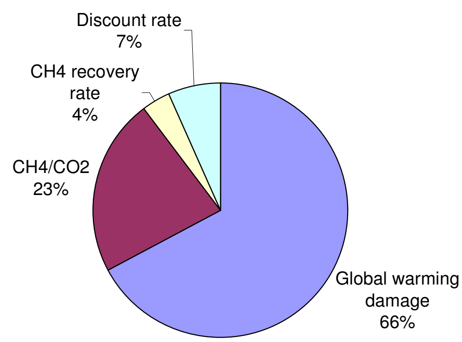


*Figure 5.5 Contributions of individual assumptions to the uncertainty of the global warming estimate.*

In comparing global warming damages of landfill across different European countries it is also interesting to focus on the $CH_4$ recovery rate alone, as this rate is probably the

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[51] The contributions to uncertainty are calculated with a decomposition technique called the Shapley decomposition method (Albrecht, Francois, & Schoors, 2002). To calculate the Shapley value of a variable, all possible orders of inclusion (permutations) of the variable in a formula are calculated and the contribution of the variable to the total effect is calculated as the average contribution over all permutations. The total effect is the difference between the 'low' and the 'high' estimate of

Table 5.10 (€ 54.50 – € 1.46 = € 53.04). With 4 variables, as in Figure 5.5, there are 4! = 24 permutations. According to Albrecht *et al.* (2002), the Shapley decomposition is perfect (there is no residual), symmetric (the order in which the variables are evaluated has no effect on the outcome) and there is no need for additional assumptions.

largest potential source of difference between these damages across countries.[52]
Table 5.11 shows 'best' global warming damage estimates at recovery rates of 25%, 50%, and 75%, respectively. For the Netherlands, an average recovery rate of 42.5 percent has been assumed.

*Table 5.11 Different assumptions on the efficiency of $CH_4$ recovery (% of recovery).*

| Recovery rate | Global warming damage (€/ton waste) |
|---|---|
| 25% recovery | 7.84 |
| 50% recovery | 5.23 |
| 75% recovery | 2.61 |

### 5.3.5 Land-use

In the literature, sometimes a special entry is made for the use of land by landfills. Because land is a marketable commodity, its cost would ideally be included in the private costs of a landfill and would not be labelled as an external cost. If, however, the land market is highly affected by government regulation (land use planning), as is the case in the Netherlands, some authors (Dijkgraaf & Vollebergh, 2004b) have argued that the 'shadow cost' of land use by landfills should be included in the estimate of the external costs of landfills. Conceptually, the 'shadow cost' of land use of a landfill is the land rent that would be offered by the next-best alternative land use. Dijkgraaf and Vollebergh (2004b) assume that the next-best alternative to a landfill would be residential development and they argue that the difference between the rent for land used for residential development and the rent for land used by a landfill is the appropriate measure for the 'shadow cost' of land use by landfills, and that this 'shadow cost' should be included in an estimate of external costs.

The DEFRA study (Enviros Consulting *et al.*, 2004) discusses external or 'shadow' costs of land use, but concludes that, in the British situation at least, the scarcity of land is likely to be reflected in the acquisition costs of landfill sites and that this scarcity would therefore not represent an environmental or social externality.

We also notice that the approach of Dijkgraaf and Vollebergh is not recommended in the official guidelines on cost-benefit analysis for large infrastructure projects in the Netherlands (Eygenraam *et al.*, 2000). The supplement to these guidelines on indirect costs (Elhorst *et al.*, 2004) discusses the welfare effects of changes in the legal classification of land, but does not advocate the valuation of land for infrastructural projects at high shadow costs. The supplement on the valuation of nature, water and soil (Ruijgrok *et al.*, 2004) does take into account potential negative effects on nature and biodiversity of land-using projects, but not of land use itself.

Hence, according to these guidelines the appropriate external cost of land use would be loss of positive externalities, such as nature, biodiversity or landscape. We do not directly see a direct correlation between the monetary value of the loss of these positive externalities and the 'shadow costs' of Dijkgraaf and Vollebergh. It should be kept in

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[52] Another source of difference is the composition of the waste, i.e., the biodegradable fraction of the waste. We have no information on the variation of this fraction across EU countries.

mind, however, that landfilling can indeed nullify positive externalities of alternative land uses. The value of this loss would be highly location-specific. We may assume that the *marginal* landfill would not be planned in the most attractive location. Things would change, however, if the number of landfills would increase significantly, so that more attractive locations would have to be sacrificed.

We included Dijkgraaf and Vollebergh's shadow cost of land use of € 17.88 per tonne of waste in our 'high' estimate of external costs, while we assume zero *external* costs in our 'low' and 'best' estimates.[53]

### 5.3.6 Other effects

The literature mentions several other external effects of landfilling, including water pollution, $NO_x$ emissions, and transport-related externalities.

#### Water pollution

In the literature, seepage of contaminants to ground and surface water is often mentioned as one of the external effects of landfilling. In this study we assume that these effects are negligible in the Netherlands, due to strict environmental standards for landfills.

#### $NO_x$

Because of the use of recovered landfill gas in gas engines to produce electricity, and because of flaring of landfill gas, $NO_x$ is emitted to the atmosphere. The volume of $NO_x$ emissions from landfills is therefore inversely related to the recovery rate of landfill gas. Dijkgraaf and Vollebergh estimate the costs of $NO_x$ emissions to be € 0.80 per tonne of waste, in case of a recovery rate of 77 percent. Linear extrapolation of this cost to recovery rates between 40 and 45 percent and applying our estimate of $NO_x$ damage (€ 3,878 per kg, see Table 5.2) results in a cost estimate between € 0.49 and € 0.55 per tonne of waste, with a 'best' estimate (at a 42.5% recovery rate) of € 0.52 per tonne of waste.

#### Transport-related externalities

On the basis of the standard values presented in Table 5.8 and an average transport intensity of 15 kilometres for each ton of landfilled waste, it can be estimated that the total external costs of transport-related activities amounts to € 1.25 per tonne of waste. Health effects are the main contributor to the transport-related externalities of landfilled waste (see Table 5.12).

### 5.3.7 Avoided emissions by power generation

A final external effect of landfills may be a *positive* external effect, as the electricity generated by recovered landfill gas might displace a certain volume of conventionally generated power and its associated externalities. Dijkgraaf and Vollebergh (2004b) assume that their 'best-practice' landfills produce 112 kWh of electricity (and no heat) per tonne of waste. If one makes the assumption that the additional supply of electricity from

[53] The private costs of landfills do, of course, include the private costs of land.

landfill gas has no effect on the market price of electricity, and that the only market effect is a substitution of the supply of electricity away from conventional power plants towards landfills, the avoided externalities of conventional power supply may be attributed to the landfill sector.

Table 5.13 shows how the avoided externalities of conventional power production are calculated (it is based on Dijkgraaf and Vollebergh, 2004b). Table 5.13 includes nine pollutants of conventional power supply, and shows how much of their emissions is avoided per ton of waste (kg/ton), the unit prices of these emissions (€/kg) in our 'best', 'low', and 'high' estimates, and, finally, the avoided pollution externalities per tonne of waste (€). The physical coefficients of the avoided emissions are based on the average power plant operating in the Netherlands, taking into account the fuel mix of Dutch power supply (CE, 1996). The row labelled 'subtotal' gives total avoided externalities per tonne of waste in the 'best-practice' landfill. The recovery ratio (average recovery rate/best-practice recovery rate) is used to account for the alternative landfill gas recovery rates we used for our 'best', 'low', and 'high' estimates. Finally, the bottom row presents the values of avoided externalities for the 'best', 'low' and 'high' estimates of landfill externalities per tonne of waste (€/t).

Note that the assumption of no market effects of the extra supply of landfill gas electricity is rather stark. One would need an economic model of the power sector to accurately assess the first- and second-order effects on emissions of pollutants of the additional supply of electricity from landfill gas.

### 5.3.8 Aggregation of external costs

Table 5.14 summarises our estimates of the external costs of landfill. Table 5.14 distinguishes between external costs of greenhouse gas emissions, other environmental pollution, land use effects, health effects, and disamenity effects. For these individual externalities, a range of marginal damage costs per tonne of waste is presented, including a 'best' estimate, and 'low' and 'high' estimates. The 'best' estimate of marginal external costs of landfilled waste is approximately € 10 per tonne of waste. The 'low' estimate is € 7 per tonne of waste, while the 'high' estimate is t € 80 per tonne of waste.

*Table 5.12 Transport related externalities for landfilling (€/ton of landfilled waste).*

| Effect | Landfilling |
|---|---|
| Climate change ($CO_2$) | 0.01920 |
| Health & materials (CO) | 0.00001 |
| Health & materials ($SO_2$) | 0.01497 |
| Health & materials ($NO_X$) | 0.10906 |
| Health & materials (HC) | 0.00329 |
| Health & materials ($PM_{10}$) | 0.92197 |
| Accidents (Serious injuries) | 0.00045 |
| Accidents (Mortality) | 0.01410 |
| Congestion | 0.17063 |
| Total transport-related externalities | 1.25367 |

The most important external effect of landfilling is climate change (i.e. 41% of the total external costs). Moreover, climate change is also the most uncertain effect. The second most important externality is the disamenity effect accounting for 34% of the total external costs. Taking account of 12% of the external effects, the third most important effect are transport related externalities.

Table 5.14 also presents (as negative entries) the maximum possible avoidance of environmental damage costs from the conventional power sector because of the production of electricity from landfill gas. By subtracting these avoided effects from the negative externalities, the net external costs of landfilling in the Netherlands result. The 'best' estimate is € 9.04 per ton of municipal solid waste.

*Table5.13 Avoided externalities from displaced power production.*

| Pollutant | | Best | | Low | | High | |
|---|---|---|---|---|---|---|---|
| | *Kg/ton* | *€/kg* | *€* | *€/kg* | *€* | *€/kg* | *€* |
| $CO_2$ | 66.7 | 0.01 | 0.67 | 0.005 | 0.33 | 0.08 | 5.34 |
| $CH_4$ | 0.4 | 0.30 | 0.12 | 0.08 | 0.03 | 2.4 | 0.96 |
| $SO_2$ | 0.05 | 7.508 | 0.38 | 7.508 | 0.38 | 7.508 | 0.38 |
| $NO_x$ | 0.09 | 3.878 | 0.35 | 3.878 | 0.35 | 3.878 | 0.35 |
| $PM_{10}$ | 0.02 | 26.765 | 0.54 | 26.765 | 0.54 | 26.765 | 0.54 |
| Bottom ash* | 0.93 | 0.009 | 0.01 | 0.006 | 0.01 | 0.075 | 0.08 |
| Fly ash | 0 | 1.136 | 0 | 1.136 | 0 | 1.136 | 0 |
| Plaster | 0 | 0.009 | 0 | 0.006 | 0 | 0.075 | 0 |
| Mine waste | 10.71 | 0 | 0 | 0 | 0 | 0 | 0 |
| Nuclear waste | 0 | 1.136 | 0 | 1.136 | 0 | 1.136 | 0 |
| *Subtotal (€/t)* | | | *2.06* | | *1.63* | | *7.64* |
| Recovery ratio | | | 42.5/77 | | 40/77 | | 45/77 |
| *Total (€/t)* | | | *1.14* | | *0.85* | | *4.46* |

* External costs of bottom ash is valued at the average external costs of landfilling (see Section 5.3.8).

Source: Dijkgraaf and Vollebergh (2004b) and Table 5.2.

*Table 5.14 External costs of landfilling one tonne of waste (€/ton).*

| Externality | Best estimate | Low estimate | High estimate |
|---|---|---|---|
| Greenhouse gas emissions ($CH_4$) | 4.21 | 1.46 | 54.50 |
| Other environmental pollution ($NO_x$) | 0.52 | 0.49 | 0.55 |
| Transport-related externalities | 1.25 | 1.25 | 1.25 |
| Land use | 0.00 | 0.00 | 17.88 |
| Health effects | 0.70 | 0.09 | 1.30 |
| Disamenity costs | 3.50 | 3.50 | 3.80 |
| *Subtotal* | *10.18* | *6.79* | *79.28* |
| Avoided externalities from the power sector | -1.14 | -0.85 | -4.46 |
| *Total* | *9.04* | *5.94* | *74.82* |

## 5.4 Incineration

### 5.4.1 Introduction

The Netherlands has 12 incinerators which in total process around 5 million tons of solid waste per annum. The incinerators are strategically distributed across the country so as to minimise transport costs and impacts (see Figure 5.6). The incineration capacity in the Netherlands grew substantially in the 1980s but has been relatively stable in the last decade. Some expansion of the incineration capacity is expected in the coming years.[54]

To estimate the external costs of waste incineration in the Netherlands, elaborate calculations have been conducted. For disamenity effects and global warming, these estimates are based on specific calculations for the Netherlands as a whole. For other effects, such as air pollution related externalities, calculations are made for one specific incinerator after which these results are extrapolated to the Netherlands in general. The waste incinerator in Alkmaar was found to be a good representative for waste incineration in the Netherlands, both in terms of capacity and environmental performance.

### 5.4.2 External effects of incineration in the Netherlands

Table 5.15 summarises the main characteristics of the Alkmaar incinerator in the North of Holland. The nominal installed capacity for the incineration plant, which has a life span of 15 years, will be 486,000 t/a, consisting of 3 incineration lines. The maximum capacity for generating power is 42 $MW_e$ of which 7 $MW_e$ is used internally.

[54] The incinerators in Alkmaar and Amsterdam are currently expanding their capacity with 650,000 tons. The incinerators in Hengelo, Wijster, Rijnmond, Moerdijk and Roosendaal have long term plans to expand their capacity with 1.5 million tons (Website AOO 2005).

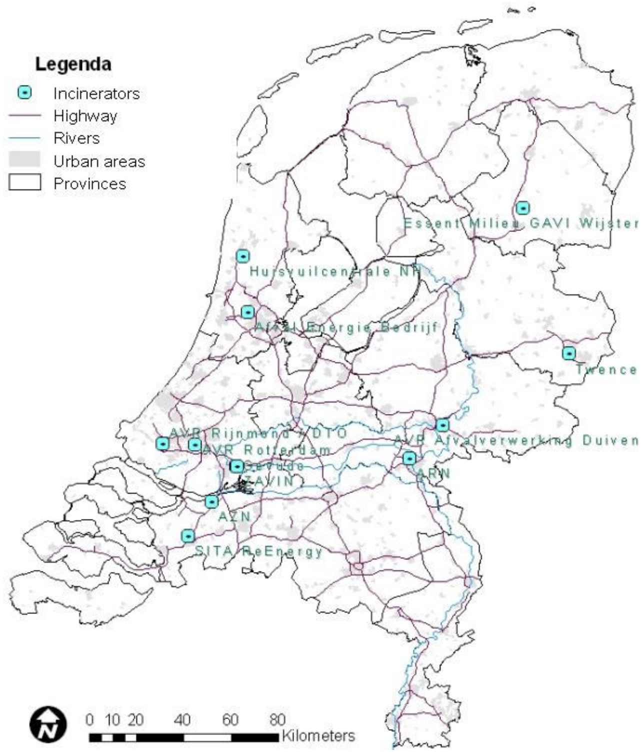


*Figure 5.6 Location of waste incinerators in the Netherlands.*

This combined heat and power generating installation is equipped with modern emission control, including dioxins and $NO_x$ reduction, according to the national legislation, which is stricter than the EU legislation. Both slag and wastewater emissions are very low as most of these materials are recycled.

Stack height has a substantial impact on the concentration of air pollutants in the short distance. Moreover, the lower the temperature of the flue gases, the more concentrated the pollutants are in the local range (Dorland *et al.* 1997; 168). The incinerator in Alkmaar has a stack of 85 meters and flue gases of 170 °C.

Finally, incineration involves the transport of various types of materials and products. These materials are assumed to be transported with a standard diesel truck. The main assumptions regarding transport were shown in Table 5.5.

In summary, the external effects resulting from the incinerator include:

- Air pollution causing increased local and regional health damage;
- $SO_2$ and $NO_x$ emissions cause reduced agricultural yield, damage to ecosystems and damage to monuments and materials;
- Transport and operation increase the risk of accidents;
- An increase in vehicle movement creates additional congestion;
- House prices are expected to decline due to residents' concern for the health effect of toxic emissions and visual intrusion from the incinerator.
- The greenhouse gases resulting from incineration are considered climate neutral. Only transport related emissions are included;
- The generation of heat and electricity prevents emissions from other energy generating sources.

The external costs in this case study are valued in monetary terms using three methodological tools. First, the impact pathway methodology (i.e. ExternE) is used to determine the public health costs, and damage to materials, buildings, crops, forests, occupational health and accidents. Second is the approach from Brisson and Pearce (1995) by which the disamenity costs of incinerators and landfills are estimated. Third, the costs of enhanced traffic congestion are determined by using values of Newbery (1992).

### 5.4.3 Health

In calculating the health damage of incineration in the Netherlands, the underlying assumption regarding the valuation of mortality, play an important role. As explained in Section 5.2, mortality can be valued in using the 'Years of Life Lost (YOLL)' approach or the 'Value of Statistical Life (VSL)' approach. To avoid a conceptual debate, both approaches have been used in the calculations. The core estimates for mortality, as shown in the first column of Table 5.16, are obtained with the YOLL approach. The sensitivity estimates for mortality impacts, shown in the second column of Table 5.16, are obtained with the VSL approach. The consequences of the choice between VSL or YOLL for valuing air emission related human health impacts is clearly demonstrated. Clearly, the VSL scenario gives much higher estimates.

*Table 5.15 Details of the incinerator in Alkmaar.*

| Technical data | Level | unit |
|---|---|---|
| Waste capacity | 486,000 | tonne/annum |
| Waste operation | 465,000 | tonne/annum |
| Full load operation time electricity | 8,117 | hours/annum |
| Gross electricity production | 42 | MWe |
| Electricity sent out | 35 | MWe |
| Stack parameters | 3 | stacks |
| Stack height | 85 | meter |
| Inside stack diameter | 1.8 | meter |
| Flue gas temperature | 170 | °C |
| Water consumption | 15,128 | $m^3$/annum |
| Construction period | 30 | months |
| Outputs | | |
| *Solid output* | | |
| - Slag | 120,267 | tonne/annum |
| - Fly ash | 14,566 | tonne/annum |
| - Recycling rate of solid output | 90 | % |
| *Water emissions* | | |
| - Temperature rise at discharge point | 0 | $^o$C |
| - Volume | 0 | $m^3$/s |
| *Air pollutant emissions* | | |
| - $SO_2$ | 5 | mg/$Nm^3$ |
| - NOx | 46 | mg/$Nm^3$ |
| - PM10 | 0.3 | mg/$Nm^3$ |
| - CO | 15 | mg/$Nm^3$ |
| - As | - | µg/$Nm^3$ |
| - Cd | < 4 * | µg/$Nm^3$ |
| - Cr | < 3 ** | µg/$Nm^3$ |
| - Hg | < 1 | µg/$Nm^3$ |
| - Ni | - | µg/$Nm^3$ |
| - PAH | < 0.5 | µg/$Nm^3$ |
| - Pb | - | mg/$Nm^3$ |
| - PCB | 0.012 | µg/$Nm^3$ |
| - PCDD/PCDF | < 1 | pg/$m^3$ |

Source: compiled from Project Appraisal Documents of the EIB (Van Beukering et al. 1998).

* ‘<‘ means ‘smaller than’. The value without the ‘<‘ sign has been used in the analysis.

** for Western European plant not only Cr but sum of Sb, Pb, Cr, Cu, Mn, V, Sn, As, Co, Ni, Se and Te.

*Table 5.16 Health effect of incineration in the Netherlands (€/ton of municipal waste).*

| | Health damage (YOLL) | Health damage (VSL) |
|---|---|---|
| *Mortality* | | |
| - $PM_{10}$ | 0.0118 | 0.0434 |
| - $SO_2$ [a] | 0.0697 | 0.2425 |
| - NOx [b] | 0.8066 | 2.9653 |
| - NOx (via ozone) | 0.0973 | 3.4589 |
| *Morbidity* | | |
| - $PM_{10}$, $SO_2$ [a] and NOx [b] | 0.1350 | 0.1350 |
| - NOx (via ozone) | 0.1728 | 0.1728 |
| *Occupational health* | | |
| - Accumulated | 0.0745 | 0.0745 |
| *Total health damage* | *1.3678* | *7.0925* |

[a] Mainly impacts due to sulphates formed from $SO_2$ in the atmosphere and direct $SO_2$ impacts.
[b] Mainly impacts due to nitrates formed from NOx in the atmosphere.

### 5.4.4 Materials and crops

The damage cost of atmospheric emissions from incineration in the Netherlands on materials and agricultural crops is shown in 5.17. Similar to health damage, $NO_x$ is the main contributor to the external costs of materials and crops. Still, the level of the damage is limited as compared to the health damages caused by air pollution.

### 5.4.5 Disamenity

Disamenity from waste incineration may be caused by various impacts. The trucks, transporting waste to and from the sites may cause noise externalities. Additionally, the incinerator may emit annoying odours. Also visual pollution results from incineration. An important factor for incineration is the increased fear for health risk by people living in its vicinity.

*Table 5.17 Material and crop damage caused by air pollution from incineration in the Netherlands (€/ton of municipal waste).*

| | Crop & material damage |
|---|---|
| *Crops* | |
| - $SO_2$ | 0.0001 |
| - NOx (via ozone) | 0.0826 |
| *Materials* | |
| - Monuments (through $SO_2$) | 0.0017 |
| - Other buildings (through $SO_2$) | 0.0420 |
| *Total damage* | *0.1264* |

There is only one relevant study that explicitly determines the disamenity impacts of incinerators. Kiel and McClain (1995) conducted a hedonic pricing study in Massachussets, USA. They find that starting at 5.5 kilometre from the plant, the house price drops by approximately € 9,500 (2005 prices) with every kilometre approaching the incinerator.

Given these findings, we assume that the disamenity effects differ between landfilling and incineration. First, the reduction in house prices seems to be more pronounced with incineration. This may be due to the fact that incineration is mainly disliked because of the perception of air pollution. In the Netherlands, since the negative publicity of the emissions of the highly toxic dioxins in the early 1990s, people are more reluctant to live near an incinerator. Even if the legal standards are met, the fear will not disappear immediately. Second, due to the importance of air emissions and the height of the stack, the impact area of an incinerator is significantly larger than the area affected by a landfill site. Therefore, the area of affected houses around the landfill site is limited to a buffer of 1 kilometre around the site. The impact area of incinerators in the Netherlands is assumed to reach as far as 5 kilometres from the actual site.

Our estimation of the disamenity effects in the Netherlands starts with the estimation of the number of residential houses within the radius of 5.5 kilometres of the plant. This radius is subdivided into zones of 1 kilometre (see as an example Figure 5.7). Next, through a GIS analysis, the house density has been determined for all active 11 incinerators in the Netherlands. Five buffer zones of 1 kilometre have been drawn around each incinerator, after which the total number of residential sites was counted. The GIS procedure has been elaborately described in the Appendix II. The main outcome of the GIS analysis is shown in Table 5.18.

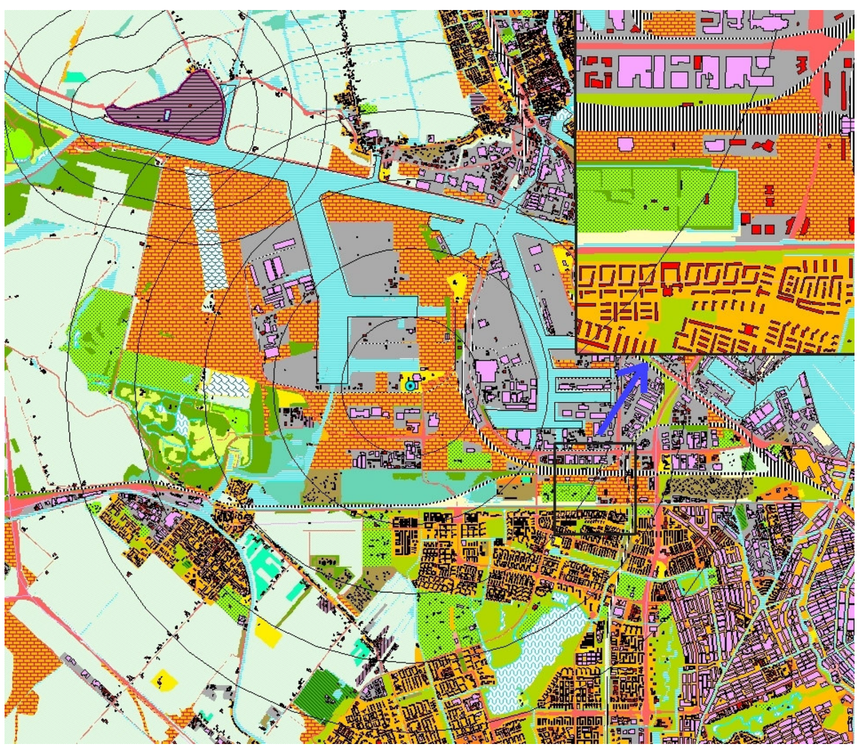

*Figure 5.7 Five respective 1-kilometre zones indicating the different levels of disamenity impact around the waste incinerator in West-Amsterdam. Red and pink polygons represent residential and commercial buildings, respectively.*

*Table 5.18 Number and density of houses located within 5 km from an incinerator in the Netherlands.*

| Impacted area | Number of houses | House density |
|---|---|---|
| *Zone* | *#* | *#/km²* |
| 1000 meter | 15,136 | 430 |
| 2000 meter | 50,534 | 363 |
| 3000 meter | 158,485 | 508 |
| 4000 meter | 260,714 | 471 |
| 5000 meter | 382,813 | 443 |
| Total | 867,682 | 430 |

Note: The house density for the 1000-meter buffer is higher than the one for the 2000-meter buffer.This illogical result is explained by the fact that one incinerator (i.e. Afvalverwerking Rotterdam) has an exceptionally high density in the 1000-meter zone, thereby accounting for 94% (14,181 of the 15,136 houses) of the houses in this zone in the Netherlands.

For landfill we determined disamenity costs of € 3.5 to € 3.8 per ton of landfilled waste (see Section 5.3.3). With 4,700,000 tons landfilled per year, this implies a value of between € 16.5 million and € 17.9 million for the 1 kilometre buffer zone. Next, we assume that the same value also exists for the same zone around the incinerators. This is reflected in the 1st row of.Table 5.19. The difference with the incinerator is that the impact zone stretches as far as 5 kilometres. As shown in the 3rd and 5th column, the impact on the house price diminishes in a linear fashion with the distance from the incinerator. By multiplying the number of houses in each zone with the house price reduction, the total value reduction can be measured (i.e. € 45.4 and € 49.3 million per year). This is converted to disamenity value per ton of waste by dividing the above value reduction by the total amount of waste incinerated (i.e. 5 million tons). As a result, the minimum and maximum disamenity value are determined at € 9 and € 10 per ton of incinerated waste.

*Table 5.19 Calculation of the disamenity value of waste incineration in the Netherlands.*

| Impact area | Number houses | House price reduction | Value reduction | House price reduction | Value reduction |
|---|---|---|---|---|---|
| Zone | # | €/house | million € | €/house | million € |
| 1000 meter | 15,136 | 1,087 | 16.5 | 1,180 | 17.9 |
| 2000 meter | 50,534 | 844 | 12.8 | 916 | 13.9 |
| 3000 meter | 158,485 | 601 | 9.1 | 652 | 9.9 |
| 4000 meter | 260,714 | 357 | 5.4 | 388 | 5.9 |
| 5000 meter | 382,813 | 114 | 1.7 | 124 | 1.9 |
| Total | 867,682 | | 45.4 | | 49.3 |
| Per ton of waste (€) | | | 9.1 | | 9.9 |

### 5.4.6 Climate change

Our general approach in estimating the effects of incineration on climate change is to estimate the (1) gross emissions of $CO_2$ and $N_2O$ (including emissions from transportation of waste to the incinerator, and ash from the incinerator to a landfill) and (2) $CO_2$ emissions avoided due to displaced electricity generation and increased production of steel and recycled inputs. To obtain an estimate of the net GHG emissions from incineration, we subtracted the GHG emissions avoided from the direct GHG emissions.

The carbon in municipal solid waste has two distinct origins. Some of the carbon in the waste is biomass carbon (i.e. carbon in plant and animal matter that was converted from $CO_2$ in the atmosphere through photosynthesis). The remaining carbon in waste is from non-biomass sources, e.g., plastics and synthetic rubber derived from petroleum. We excluded biogenic $CO_2$ emissions from combustion of biomass but included the combustion of the non-biomass components in the solid waste.

A study for the USEPA (ICF 1997) shows that under the above mentioned assumptions, the gross GHG emissions per ton of incinerated municipal solid waste (MSW) are equal to 0.12 metric tons of carbon equivalent (MTCE) (see 5.20). Avoided energy and material production leads to a decline in emissions of 0.08 MTCE. Assuming that the waste composition in the USA is similar to the Netherlands, the net GHG emission from incinerating one ton of MSW in the Netherlands is estimated at 0.04 MTCE, which is equivalent to 0.011 ton $CO_2$.

*Table 5.20 Quantification of net GHG emissions from MSW incineration.*

| | $CO_2$ from incineration *MTCE/ton* | $N_2O$ from incineration *MTCE/ton* | $CO_2$ from transport *MTCE/ton* | GHG Emissions *MTCE/ton* | $CO_2$ equivalent *MT CO2/ton* |
|---|---|---|---|---|---|
| Gross emissions | 0.10 | 0.01 | 0.01 | 0.12 | 0.033 |
| Avoided electricity generation | - 0.07 | | | - 0.07 | -0.019 |
| Avoided material | - 0.01 | | | - 0.01 | -0.03 |
| Net GHG emissions | | | | 0.04 | 0.011 |

Source: ICF, 1997, p.86-91.

Table 5.21 summarizes the assumptions that are used in estimating global warming damages from incineration in the Netherlands. The table reports ‘low’ and ‘high’ estimates of various assumptions and also a ‘best’ estimate. The ‘best’ estimate of global warming damages of incineration is based on the ‘best’ estimates of the underlying assumptions. This ‘best’ estimate is € 0.11 per ton of waste.

*Table 5.21 Estimates of global warming damages of incineration in the Netherlands.*

| Assumption | ‘Best’ | Low | High |
|---|---|---|---|
| Global warming damage €/t$CO_2$ | 10 | 5 | 80 |
| Net $CO_2$ equivalent emissions | 0.011 | 0.011 | 0.011 |
| Global warming damage €/ton waste | 0.11 | 0.055 | 0.88 |

### 5.4.7 Other effects

#### Transport-related externalities

On the basis of the standard values presented in Table 5.8 and an average transport intensity of 20 kilometres for each ton of incinerated waste, it can be estimated that the total external costs of transport-related activities amounts to € 1.67. Health effects are the main contributor to the transport-related externalities of incinerated waste.

### Solid and chemical waste residues

Figure 5.8 shows the treatment and disposal of solid residues of incineration, as practiced in the Netherlands. Solid residues consist of bottom ash and the more toxic residues from flue gas treatment. Bottom ash is produced from the combustion process and comprises around 20-30% of the mass of the waste treated at an incinerator. It is possible to recover materials such as iron and aluminium from bottom ash. The remainder of the bottom ash is generally disposed of to a to a controlled landfill. Under some conditions, however, the utilisation of bottom ash in road base materials is allowed.

Residues generated through the process of flue gas treatment are typically hazardous. These residues include fly ash and residues obtained from acid gas cleaning. Dutch incinerators generate around 20 kg of fly ash and 1-20 kg of acid gas cleaning residue per ton of waste incinerated (Hjelmar, 1998). Flue gas residues are problematic due to the high concentration of heavy metals and are therefore disposed of in controlled landfill sites designed for
hazardous waste, either separately (monofill) or, less frequently, combined with bottom ash (landfill).

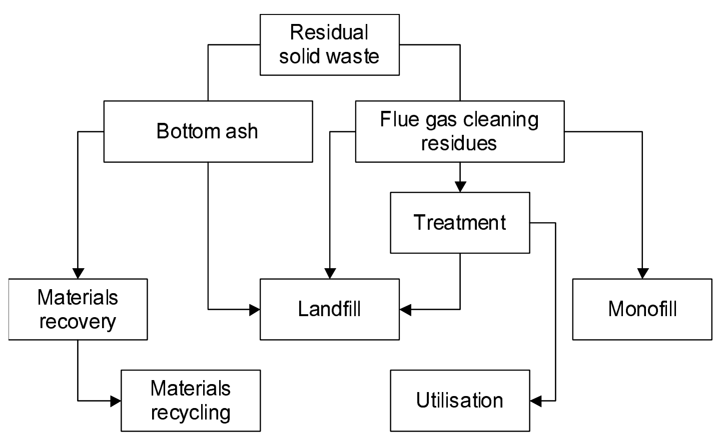


*Figure 5.8 Potential treatment and disposal routes for solid waste residues from MSW incineration plants.*

Source: COWI, 2000, p.23.

In our calculations we assume that the incineration of one ton of waste in the average Dutch plant generates 258 kg of bottom ash (Van Beukering *et al.* 1998) and 7.8 kg of chemical waste (CE 1996). The relatively harmless bottom ash which in the Netherlands is generally used as road and construction material, in the 'best' scenario is valued at zero external costs. In the 'high' cost scenario, the cost of bottom ash is valued at the external costs of landfilling in the Netherlands, exclusive of the costs of greenhouse gas emissions, because bottom ashes do not contain organic materials (see calculation in Section 5.3).

The external costs of chemical waste disposal can be based on two assumptions. On the one hand, the fact that chemical waste is hazardous may be a reason to apply high external costs. This is reflected in the estimate provided by CE (1996). On the other hand, if the chemical waste is properly stored in a controlled landfill or monofill, the hazardous content of the waste does not lead to externalities, unless accidents happen to the landfill or during the transport (Ascari *et al.*, 1995). However, no information could be found on the probability of such incidents. Therefore, we also estimate the external costs of chemical waste assuming that the risk of accidents is negligible because it has been internalised into in the overall treatment costs. In that case, we value the chemical residues at the external costs of landfilling in the Netherlands, exclusive of the costs of greenhouse gas emissions. Table 5.22 shows the overall external costs of solid waste residues from incineration.

### 5.4.8 Avoided emissions by power generation

Similar to landfills, incinerators in the Netherlands generate *positive* externalities through the production of heat and electricity. This implies that processing waste through incineration displaces the conventional generation of electricity and heat with its related negative externalities. We tap several sources of background data to estimate the size of these avoided externalities.

First, Dijkgraaf and Vollebergh (2004b) assume that their 'best-practice' incinerators generate 580 kWh of electricity and 299 kWh of heat per ton of waste. Previous calculations in the ExternE study assume an average output equivalent of 614 kWh per ton of waste. We adopt the latter estimates for the calculation of the level of the avoided emissions. Second, the value associated with each air pollutant are applied from both CE (1996) and ExternE (EC, 1995) to determine an upper and a lower estimate of the avoided externalities. As with landfills, we assume that the additional supply of electricity from incineration has no effect on the market price of electricity, and that the only market effect is a substitution of the supply of electricity away from conventional power plants towards incinerators, the avoided externalities of conventional power supply may be attributed to the incineration sector. It is assumed that the displaced energy represents the average fuel mix for power plants in the Netherlands.

*Table 5.22 External costs of solid waste residues of incineration (€/ton).*

| Waste residues | Quantity | Costs per ton of pollutant | | | External costs per ton of waste | | |
|---|---|---|---|---|---|---|---|
| | | *'best'* | *'low'* | *'high'* | *'best'* | *'low'* | *'high'* |
| *Unit* | *kg/ton* | *€/ton* | *€/ton* | *€/ton* | *€/ton* | *€/ton* | *€/ton* |
| Bottom ash | 259 | 0 | 0 | 0.02 | 0 | 0 | 5.18 |
| Chemical waste | 21.8 | 0.005 | 0.004 | 0.02 | 0.11 | 0.09 | 0.44 |
| External costs solid residues | | | | | 0.11 | 0.09 | 5.62 |

Table 5.23 includes eight pollutants of conventional power supply, and shows how much of their emissions is avoided per ton of waste (kg/ton), the unit prices of these emissions (€/kg) in our 'best', 'low' and 'high' estimates, and, finally, the avoided pollution externalities per ton of waste (€).

*Table 5.23 External cost estimates for avoided emissions from incineration in the Netherlands.*

| Pollutant | Pollutant | Best estimate | | Low estimate | | High estimate | |
|---|---|---|---|---|---|---|---|
| | kg/ton waste | €/kg | €/ton waste | €/kg | €/ton waste | €/kg | €/ton waste |
| $CO_2$ | 365 | 0 | 0 | 0.00[a] | 0.00 | 0 | 0.00 |
| $CH_4$ | 2.192 | 0.30 | 0.67 | 0.008 | 0.02 | 2.4 | 5.26 |
| $SO_2$ | 0.274 | 7.508 | 2.06 | 7.508 | 2.06 | 7.508 | 2.06 |
| $NO_x$ | 0.493 | 3.878 | 1.91 | 3.878 | 1.91 | 3.878 | 1.91 |
| $PM_{10}$ | 0.11 | 26.765 | 2.94 | 26.765 | 2.94 | 26.765 | 2.94 |
| Bottom ash | 5.096 | 0.009 | 0.04 | 0.006 | 0.03 | 0.075 | 0.38 |
| Fly ash | 0 | 1.136 | 0 | 1.136 | 0.00 | 1.136 | 0.00 |
| Mine waste | 58.686 | 0 | 0 | 0 | 0.00 | 0 | 0.00 |
| Total | | | 7.63 | | 6.96 | | 12.56 |

[a] The external benefits of climate change related to incineration are addressed in the previous Chapter and therefore not accounted for in this estimate of avoided emissions.

### 5.4.9 Aggregation of external costs of incineration

The estimates of the external costs of incineration are summarised in Table 5.24, which distinguishes between external costs of health, materials, agriculture, disamenity, congestion, and climate change. For these individual externalities, a range of marginal damage costs per tonne of waste is presented, including a 'best' estimate, and 'low' and 'high' estimates. The 'best' estimate of marginal external costs of incinerated waste (not taking into account the avoided environmental impacts of energy production) is approximately € 18 per ton of waste. The 'low' estimate is € 12 per ton of waste, while the 'high' estimate is € 25 per ton of waste.

*Table 5.24 Aggregation of external costs of incineration in the Netherlands (€/ton).*

| Effect | Best estimate | Low estimate | High estimate |
|---|---|---|---|
| Health | 7.09 | 1.37 | 7.09 |
| Materials & agriculture | 0.13 | 0.13 | 0.13 |
| Disamenity | 9.09 | 9.09 | 9.87 |
| Transport-related | 1.67 | 1.67 | 1.67 |
| Solid waste | 0.11 | 0.09 | 5.62 |
| Climate change | 0.11 | 0.06 | 0.88 |
| *Subtotal* | *18.20* | *12.41* | *25.26* |
| Displaced effects | -7.63 | -6.96 | -12.56 |
| *Total* | *10.57* | *5.45* | *12.70* |

The most important externality of incineration is the disamenity effect (i.e. 50% of the total (gross) external costs). The second most important externality is the negative impact on health through air emissions. Health effects account for 39% of the external costs. Taking account of 9% of the external effects, the third most important effect is transport-related externalities. As opposed to the situation with landfilling, the importance of the effects of climate change caused by incineration is minimal. Climate change effects contribute only 1% to the overall external costs.

Table 5.24 also presents (as negative entries) the maximum possible avoidance of environmental damage costs from the conventional power sector because of the production of electricity and heat from incineration. By subtracting these avoided effects from the negative externalities, the net external costs of incineration in the Netherlands result. The 'best' estimate is € 10.57 per ton of municipal solid waste.

## 5.5 Cost benefit analysis

### 5.5.1 International comparison of external costs

The external costs of landfill and incineration have been the subject of a relatively large number of studies in recent years. Most of these studies focus only on one specific externality. Only a few studies attempt to aggregate a number of the most relevant environmental effects, thereby allowing for a fair comparison, internationally. In this Chapter, we will compare these studies with our findings.

The first study we address is a study by Ascari *et al.* (1995) that has been prepared for the ExternE Project in which a comparison is made between incineration and landfilling in both rural and urban areas. The study takes into account the effects on health (i.e. morbidity, mortality), agriculture and materials, climate change and accidents. The study excludes several potentially important categories of externalities, such as the disamenity impact of incinerators and landfills, and the effect on soil and groundwater. The study therefore warns for using these results for the setting of levies and subsidies to waste disposal technologies until a number of these missing values are properly assessed.

The impact of the location (i.e. urban and rural) on the level of externalities is clearly visible in Figure 5.9. The external cost of incineration in Italy varies between € 16 and € 27 per tonne incinerated. The external costs of landfilling are substantially lower, varying between € 3 and € 4 per ton of landfilled waste.

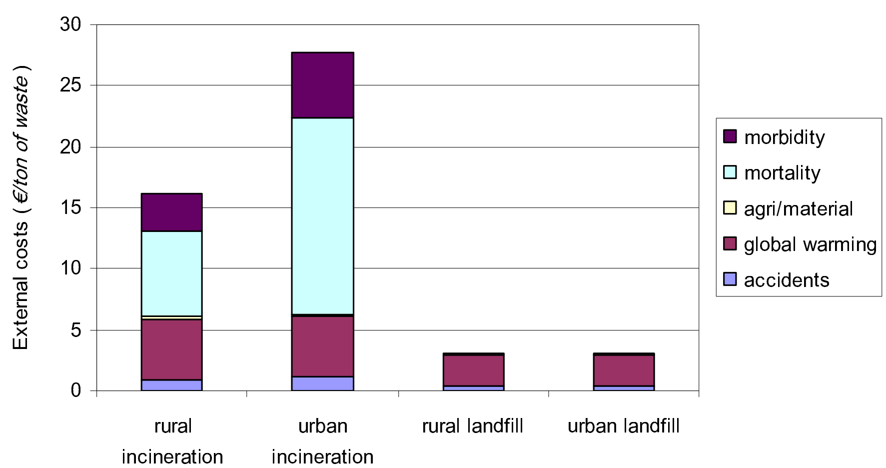


*Figure 5.9 External costs of incineration and landfilling in Italy (€/ton).*

Source: Ascari *et al.* (1995).

The second study that quantifies the external costs of landfilling and incineration is prepared for a hypothetical location in Europe (COWI 2000). The examples have been designed to reflect respectively old obsolete and new modern waste disposal plants for both landfilling and incineration. The examples include the external costs of climate change, conventional air pollutants and some airborne toxic substances, leachate and disamenity effects. The calculations include external benefits in the form of displaced emissions from energy recovery but exclude externalities related to transport of waste and residues.

Table 5.25 summarises the main results of the COWI study. The first typical finding of the analysis is that the modern incinerator actually generates external benefits, rather than external costs. Although the gross external costs account for € 28 per ton of incinerated waste, the displaced energy generated external benefits of € 70, thereby leading to a net gain of €42 per ton of incinerated waste. With a net externality of € 37, the obsolete incinerator is much less environmentally friendly. Both the modern and old landfill, perform better than the old incinerator. The main contributors to the external effects are climate change for landfilling and air-related health effects for incineration.

*Table 5.25 External cost of incineration and landfilling in Europe (€/ton).*

| | Incineration (85% energy recovery) | Incineration (25% energy recovery) | Landfilling (BAT) | Landfilling (old) |
|---|---|---|---|---|
| Global warming | 0.77 | 0.77 | 4.91 | 8.19 |
| Air emissions | 19.82 | 50.08 | 0.12 | 0.00 |
| Water pollution / leachate | 0.00 | 0.00 | 0.00 | 1.52 |
| Disamenity | 7.50 | 7.50 | 10.01 | 10.01 |
| *Total external costs* | *28.09* | *58.36* | *15.04* | *19.72* |
| Energy savings | -69.66 | -20.98 | -4.03 | 0.00 |
| *Net external costs* | *-41.56* | *37.37* | *11.01* | *19.72* |

Source: COWI (2000), p.60 & p.63.

The final study that compares landfilling and incineration is from the Netherlands itself (Dijkgraaf and Vollebergh, 2004b). On the basis of existing information on technical properties of installations, emission levels and monetary estimates, the external costs of the average incinerator and landfill site have been estimated (see Table 5. 6). Taking into account the external benefits from the displaced energy and materials, incineration generates slightly less external costs than landfilling in the Netherlands (i.e. € 18 vs € 22).

The estimates of the externalities of the incinerator are higher than the previous studies. One reason for this difference is the high value for chemical waste. Whether this is justified is uncertain. As mentioned earlier, most studies assume that chemical waste is stored properly and therefore does not form a serious environmental threat. The landfill estimates also have an uncommon component: land use. As discussed in Section 5.3, this category is usually not considered as an external effect.

*Table 5. 26 External costs of incineration and landfilling in the Netherlands (€/ton).*

| Externality | Landfilling | Incineration |
|---|---|---|
| Air emissions | 5.84 | 17.26 |
| Water pollution | 0.00 | 0.00 |
| Chemical waste | 2.63 | 28.69 |
| Land use | 17.88 | 0.00 |
| *Sub total* | *26.35* | *45.95* |
| Energy savings | - 4.21 | - 22.55 |
| Material savings | 0.00 | - 5.76 |
| *Total external costs* | *22.14* | *17.64* |

Source: Dijkgraaf and Vollebergh (2004b).

### 5.5.2 Comparison of private costs

In the literature, several studies can be found that explicitly calculate the private costs of incineration. Generally, these costs are affected by:

- Costs of land acquisition;
- Scale (there are significant diseconomies of small scale);
- Plant utilisation rate;
- The level of treatment of flue gas;
- The level of treatment and disposal of ash residues;
- The efficiency of energy recovery, and the revenue received for energy delivered;
- The recovery of metals and the revenues received from this.

A rough cost breakdown for a 200,000 tonne facility is given in Table 5.27.

*Table 5.27 Examples of incineration of solid waste in Europe.*

| Cost category | Flanders | Germany |
|---|---|---|
| *Unit* | *€ per ton waste* | *€ per ton waste* |
| Capital costs | 37.08 | 63.61 |
| Operational costs (input independent) | 31.76 | 38.56 |
| Operational costs (input dependent) | 18.04 | 19.10 |
| Credits for electricity | -11.76 | -16.27 |
| *Total costs* | *75.12* | *105.00* |

Source: Hogg (2001).

The estimate of the private costs of incineration in the Netherlands used in this analysis is based on a study conducted by the Dutch Ministry of Finance (quoted by Dijkgraaf and Vollebergh (2004b)). These facts and figures accurately reflect the real costs of incineration at a state-of-the-art technology in the Dutch context. The background data on savings on the private costs of generating electricity and materials production are based on Dijkgraaf and Vollebergh (2004b). The costs of electricity production through a gas-fired power plant in the Netherlands is € 0.036 per kWh (2000 prices).

It is much more difficult to find reliable data on the true costs of landfill operations. Especially, the Dutch context is poorly administered publicly (Dijkgraaf and Vollebergh (2004b). There is a risk in transferring international estimates to the Dutch context because the Netherlands has one of the strictest legislations regarding leakage of effluent in the world. Still, considering European studies for this purpose may help to get a better idea of the range of landfill costs (see Table 5.28).

Landfill costs can typically be disaggregated into five components. To indicate the importance of each component, an example from the United Kingdom (excluding energy recovery) is added:

- Acquisition costs (e.g. UK example: € 1.56 per ton);
- Capital expenditure and development costs (e.g. UK example: € 11.46 per ton);
- Operating costs (e.g. UK example: € 10.97 per ton);
- Restoration (e.g. UK example: € 0.78 per ton);
- Aftercare costs (e.g. UK example: € 4.01 per ton).

The estimates used in our assessment are based on the only Dutch estimates available as well as expert judgements (see Table 5.29). The costs of landfilling in The Netherlands (€ 36 per tonne[55]) seem to be lower than in many other EU Member States (see Table 5.28).

*Table 5.28 Comparative costs of landfill in different member states (€/ton).*

| Country | Financial costs |
|---|---|
| Austria | 67 |
| Belgium | 45 |
| France | 31-85 |
| Germany | 20-51 |
| Italy | 52 |
| Luxemburg | 123-147 |
| United Kingdom | 28 |
| *Average* | *59* |

Source: Hogg (2001), p.64.

*Table 5.29 Private costs estimates for landfilling and incineration (€/ton).*

| Cost category | Incineration | Landfilling |
|---|---|---|
| Gross private costs | 125 | 40 |
| Displaced costs - energy | -21 | -4 |
| Displaced costs - materials | -3 | -0 |
| Net private costs | 101 | 36 |

Source: Dutch Ministry of Finance quoted by Dijkgraaf and Vollebergh (2004b). The private costs
for incineration are based on expert judgements.

### 5.5.3 Cost benefit analysis

The main purpose of the economic valuation of the externalities of incineration and landfilling in the Netherlands is two-fold. First, we want to find out whether the landfill tax is representative for the external costs of landfilling in the Netherlands. At present, landfill

55 This figure seems to be reasonably reliable. For comparison: a survey among members of the Vereniging Afvalbedrijven revealed an average landfilling tariff of € 38 per tonne in 2003 (personal communication J. Bouman, NV Afvalzorg, August 2005).

tax in the Netherlands is almost € 85 per ton. Compared to many other European countries, this is rather high. Moreover, because of the technical requirements on the management of landfill sites and restrictions on inputs, gate fees in 2003 are typically € 125 per ton. These high fees strongly discourage the disposal of municipal solid waste in landfills. The last column of Table 5.30 shows the external costs of landfilling in the Netherlands. Clearly, the ‘best’ estimate of € 10.33 per ton of landfilled waste is considerably lower than the current level of the landfill tax. As we have argued, however, the size of the external effects of landfilling is uncertain, to a large extent because of the uncertainty of future damage through global warming and of uncertainty of the extent to which the opportunity cost of land exceeds its market value. Taking these sources of uncertainty into account, the landfill tax exceeds the uncertainty range of marginal external damage by a small margin only.

Second, we want to test the validity of the fundament of the Dutch waste policies, the ‘waste hierarchy’, which identifies incineration with energy recovery as preferred to landfilling with energy recovery. In order to fulfil this objective, we estimated the social costs by adding up the private and external costs. The results are shown in Table 5.30. Taking the ‘best’ estimate as the basis of our calculation, we conclude that, at the margin, the social costs of incineration (€ 112) exceed the social costs of landfilling (€ 45) substantially. However, the range of uncertainty for the social costs of landfilling is relatively large, and the ‘high’ estimates for both types of waste management are quite close to each other.

*Table 5.30 Private, external and social costs of incineration and landfilling of municipal solid waste in the Netherlands (€/ton) .*

| *Cost category* | *Scenario* | Incineration *€/ton* | Landfilling *€/ton* |
|---|---|---|---|
| Private costs | | 101.00 | 36.00 |
| External costs | Best | 10.57 | 9.04 |
| *External costs* | *Low* | *5.28* | *5.94* |
| *External costs* | *High* | *12.65* | *74.82* |
| Social costs | Best | 111.57 | 45.04 |
| *Social costs* | *Low* | *106.28* | *41.94* |
| *Social costs* | *High* | *113.65* | *110.82* |

## 5.6 Co-incineration of waste

Co-incineration of solid waste residues in coal-fired power plants is becoming increasingly popular in the Netherlands. For example, at present 18% of plastic waste in the Netherlands is processed in cement kilns and power plants (Jaarverslag Commissie Verpakkingen, 2003). To be suitable for co-incineration processes, solid waste has to meet several criteria. First, the waste should have a high-energy value in order to meet the temperature limits of the furnaces and kilns. Second, the solid waste should not contain high concentrations of pollutants such as heavy metals to avoid violations of the existing air emission standards in the Netherlands. Third, the waste should generate a particular quality of bottom ash so that the ash can still be used in alternative applications such as the cement industry. Fourth, also specific physical conditions need to be met to allow for grinding of the materials (AOO, 2002b).

The solid waste residues that meet these criteria include A-wood, B-wood, and paper and plastics residues (PPF). Therefore, co-incineration provides only a partial solution to a particular type of solid waste in the Netherlands. Because the waste types for the various waste management processes varies widely, a straightforward comparison between incineration, landfilling and co-incineration is not fully justified. While landfilling and incineration process a random mixture of household waste, co-incineration handles specific industrial and institutional waste with high caloric value. Another constraint of comparing incineration, landfilling and co-incineration is that especially incineration and co-incineration are mutually dependent. The efficiency of incineration can improve by extracting waste with high calorific value from the overall waste flow, for example through separating waste for the purpose of co-incineration. In other words, strictly speaking incineration and co-incineration cannot be analysed independently.

Due to these fundamental limitations, we only provide an rough approximation of the external costs of co-incineration. We re-emphasize, however, that such a comparison is one of comparing 'apples and oranges'. The results should therefore only be considered as a means of improving our understanding of co-incineration as an additional waste management option, rather than as robust measure of the social desirability of this process.

Two sources explicitly report on the environmental performance of co-incineration of solid waste in coal fired power plants and cement kilns in the Netherlands. Croezen and Bergsma (2000) conduct and environmental impact assessment of the processing of waste plastics for various incineration processes such as the traditional method of incineration, gasification, and co-incineration as sub-coal in cement kilns and coal fired power plants. As shown in Table 5.31, both processing through coal fired plants and cement kilns generates better environmental scores for all environmental themes except for VOCs. In fact, according to their study, co-incineration provides an environmental improvement, rather than merely a reduction in environmental pressure. The advantages of both processes compared to conventional incineration is that they perform much better for climate change, acidification and solid waste. Unfortunately, the results of Croezen en Bergsma can not be used to determine the external monetary value of co-incineration because their method of valuing the emissions is based on a fundamentally different process.

AOO (2002b) provides the environmental impact of various incineration scenarios in the Netherlands. These scenarios involve different levels of co-incineration. Two scenarios are relevant for our study. The first is the 'integrated incineration' scenario in which landfilling and co-incineration are gradually eliminated before 2012 and the incineration capacity in the Netherlands is further expanded by introducing novel incineration techniques. The other scenario is labelled 'co-incineration' and involves the expansion of the co-incineration capacity from 0.5 million tons by 2005 to 2.5 million tons by 2012. This expansion goes at the expense of the incineration capacity. The calculated environmental impact of both scenarios, and the difference between them, is shown in Table 5.32.

*Table 5.31 Contribution of different environmental problems to total environmental indicator of three processing methods.*

| Environmental problem | Conventional incineration | Co-incineration coal fired Power plant | | Co-incineration cement kiln | |
|---|---|---|---|---|---|
| | Absolute value (*$10^{-9}$ per ton waste plastic*) | Absolute value (*$10^{-9}$ per ton waste plastic*) | Relative difference to incineration | Absolute value (*$10^{-9}$ per ton waste plastic*) | Relative difference to incineration |
| Climate change (*kg $CO_2$ –eq*) | 9.3 | -2.6 | -128% | -3.2 | -134% |
| Acidification (*kg $SO_2$ –eq*) | 1.6 | -3.4 | -313% | -0.5 | -131% |
| Euthrophication (*kg $PO_4$ –eq*) | 0.1 | -0.9 | -1000% | -0.1 | -200% |
| VOCs (*kg VOCs*) | -0.9 | 0.1 | -111% | 0 | -100% |
| Air pollution (*kg poll. env*) | 0 | -0.1 | n.a. | 0 | n.a. |
| Solid waste (*kg solid waste*) | 17.7 | 0 | -100% | 0 | -100% |
| Total | 27.8 | -6.9 | -125% | -3.8 | -114% |

Source: based on Croezen and Bergsma (2000).

*Table 5.32 Overview of main environmental impacts of the incineration and the co-incineration scenarios.*

| | Unit | Incineration | Co-incineration | Difference |
|---|---|---|---|---|
| Energy | GJ/ton | 6.2 | 3.6 | -42% |
| Waste disposal | kg/ton | 0 | 0 | n.a. |
| CO2 | kg/ton | 571 | 580 | 2% |
| NOx | kg/ton | 0.41 | 0.68 | 66% |
| NH3 | kg/ton | 0.02 | 0.011 | -45% |
| N2O | kg/ton | 0 | 0.004 | n.a. |
| CO | kg/ton | 0.15 | 0.2 | 33% |
| CH | kg/ton | 0.04 | 0.18 | 350% |
| PM10 | kg/ton | 0.0219 | 0.0172 | -21% |
| Dioxines | kg/ton | 3.6E-10 | 2.1E-10 | -42% |
| Transport | Tonkm | 60 | 91 | 52% |
| Space | $m^2$/ton year | 0.14 | 0.22 | 57% |

Source: based on AOO, 2002b.

On the basis of the above information in Table 5.32, we are able to modify the earlier monetary estimate on incineration, as presented in the first column of Table 5.33. The difference for the health estimate is based on a weighted average of NOx and PM10 emissions. The material and agricultural damage is based on NOx only. The modification of disamenity is based on the fact that co-incineration requires substantially more space. The other notable effect is the reduction in displaced energy.[56] Ultimately, the external costs in the co-incineration scenario are significantly higher than for the integrated incineration scenario.

Because both studies report opposite results, we feel the monetary estimate for co-incineration reported in Table 5.33 is not very reliable. Nevertheless, as it is the best we can get, we will use this estimate as the basis for our conclusions in the next chapter.

*Table 5.33 Aggregation of external costs of co-incineration in the Netherlands (€/ton)*

| Effect | Incineration [a] | Change [b] | Co-incineration [c] |
|---|---|---|---|
| Health | 7.09 | +48% | 10.52 |
| Materials & agriculture | 0.13 | +66% | 0.22 |
| Disamenity | 9.09 | +57% | 14.28 |
| Transport-related | 1.67 | +52% | 2.53 |
| Solid waste | 0.11 | 0 | 0.11 |
| Climate change | 0.11 | +2% | 0.11 |
| *Subtotal* | *18.20* | | *27.77* |
| Displaced effects | -7.63 | -42% | -4.43 |
| *Total* | *10.57* | | *23.34* |

[a] based on estimates from this study (see previous chapter).
[b] based on difference derived from AOO, 2002b.
[c] combination of column 1 and 2.

## 5.7 Conclusions

To wind up this chapter, we return to the research questions that it had to answer:

- What are the social costs and benefits of the following treatment options for household waste and comparable waste from firms:
- Landfilling all waste;
- The present situation;
- Terminating the landfilling of combustible waste by shifting it to waste incineration;
- Terminating the landfilling of combustible waste by shifting it to co-incineration (in power plants);
- Terminating the landfilling of combustible waste by optimizing/maximizing recycling.

A definitive answer to this question cannot be given, due to lacking and sometimes contradictory information, especially regarding co-incineration and various recycling op-

[56] One could argue that in case of co-incineration the appropriate reference situation is a 100% coal power plant, rather than the average fuel mix for Dutch power plants (because co-incineration will usually take place in coal power plants). In that case the absolute value of the ‘displaced effects’ would be somewhat higher. Nevertheless, the total external costs of co-incinaration would still be much higher than those of incineration.

tions. However, if we assume the social costs to be zero for recycling[57], while adopting the external costs estimated by AOO (2002b) for co-incineration, some preliminary estimates can be provided for the above scenarios. For this purpose we utilise the 'best' estimates of the net social costs for each option, as estimated in the preceding Chapters. Next, we need to determine the process mix in terms of physical quantities of each of the five scenarios. The most recent source of the allocation of waste management options in the Netherlands is provided by the Milieu en Natuur Planbureau in the Netherlands. The total amount of household and institutional/office waste involved is 13.788 mln ton (MNP, 2005), of which in the present situation 48% is recycled (excluding co-incineration in power plants), 38% is incinerated, 4% co-incinerated, 8% landfilled and 2% discharged (liquid waste). Because this latter category is both rather small and extremely varied in terms of external effects, we exclude the 'discharge' category in the scenario analysis. It is assumed that in principle all waste from households and comparable waste from firms is combustible.

*Table 5.34 Quantities of household and institutional/office waste processed in the Netherlands in 2003 (in 1,000 tons per year).*

| | Household waste | Institutional/office waste | Total waste | Share |
|---|---|---|---|---|
| Recycling | 4,147 | 2,498 | 6,644 | 48% |
| Incineration | 3,633 | 1,645 | 5,278 | 38% |
| Co-incineration* | 363 | 165 | 528 | 4% |
| Landfill | 519 | 556 | 1,075 | 8% |
| Discharge | 147 | 116 | 263 | 2% |
| | 8,809 | 4,979 | 13,788 | |

* Co-incineration is not provided in the original data from MNP. Therefore, we adopted the level of around half a million tons of co-incinerated waste reported by AOO (2002b). This amount was deducted from the recycled amount as reported by MNP, who apply a broad definition of recycling (i.e. 'useful applications').

Source: based on MNP 2005, Table B2.1, p.130.

The physical flows of the five scenarios are shown in the upper part of Table 5.35. The next parts of Table 5.35 show the private, external, avoided energy and the social costs, respectively. The lowest social costs are recorded in scenario 1 which hypothetically treats all the solid waste through landfilling. The second best option is the maximisation of recycling scenario (i.e. scenario 5). Yet, this result is mainly driven by the fact that the external costs of recyling are unknown and therefore are assumed to be zero. The highest social costs are recorded by scenario 4, in which co-incineration is maximised. Obviously, the figures presented here should be treated with caution, as they are based on *marginal* social costs and therefore can strictly speaking not be used to calculate *total* social costs.

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[57] The external costs of recycling can be positive or negative, depending on the type of the recycled material and on the primary material that it replaces. Similarly, the private cost of recycling can be positive or negative, depending on various specific circumstances (including technical factors and prevailing market prices). We consider any other estimate than zero to be more arbitrary.

- To what extent can the current level of the Dutch landfill tax rate be regarded as an internalisation of the environmental costs of landfilling?

The calculations made in Section 5.3 strongly suggest that the current landfill tax rate of almost € 85 per tonne is at least equal to, but probably substantially higher than the marginal external costs of landfilling. The 'best' estimate of the external costs is about € 9 per tonne, and even the 'high' estimate (€ 75 per tonne) is lower than the current tax rate.

- From a social cost-benefit perspective, what is the optimum way of waste treatment for household waste and comparable waste from firms?

A definitive answer to this question cannot be given, due to lacking and sometimes contradictory information, especially regarding co-incineration and various recycling options. Nevertheless, it is clear that the net social costs of landfilling are probably much lower than those of (co-)incineration. Some combination of landfilling (with methane recovery) and recycling might well be the strategy implying minimum net social costs. Obviously, such a strategy would be dependent on the availability of sufficient space for (new) landfill sites.

*Table 5.35 Aggregated private, external, and avoided costs of five waste management scenarios in the Netherlands (million € per year).*

| | | Scenario 1 | Scenario 2 | Scenario 3 | Scenario 4 | Scenario 5 |
|---|---|---|---|---|---|---|
| | *Unit value (€/ton)* | Landfilling all waste | Current Situation | Switch from landfill to Incineration | Switch from landfill to co-incineration | Maximize Recycling |
| *Quantities* | | *Million tons* | *Million tons* | *Million tons* | *Million tons* | *Million tons* |
| Landfilling | | 13.5 | 1.1 | 0.0 | 0.0 | 0.0 |
| Incineration | | 0.0 | 5.3 | 6.4 | 5.3 | 5.3 |
| Co-incineration | | 0.0 | 0.5 | 0.5 | 1.6 | 0.0 |
| Recycling | | 0.0 | 6.6 | 6.6 | 6.6 | 8.2 |
| Total | | 13.5 | 13.5 | 13.5 | 13.5 | 13.5 |
| *Private costs* | *€/ton* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* |
| Landfilling | 36 | 487 | 39 | 0 | 0 | 0 |
| Incineration | 101 | 0 | 533 | 642 | 533 | 533 |
| Co-incineration | 101 | 0 | 53 | 53 | 162 | 0 |
| Recycling | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | | 487 | 625 | 695 | 695 | 533 |
| External costs | *€/ton* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* |
| Landfilling | 10.18 | 138 | 11 | 0 | 0 | 0 |
| Incineration | 18.20 | 0 | 96 | 116 | 96 | 96 |
| Co-incineration | 27.77 | 0 | 15 | 15 | 45 | 0 |
| Recycling | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | | 138 | 122 | 130 | 141 | 96 |
| Avoided externalities | *€/ton* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* |
| Landfilling | -1.14 | -15 | -1 | 0 | 0 | 0 |
| Incineration | -7.63 | 0 | -40 | -48 | -40 | -40 |
| Co-incineration | -4.43 | 0 | -2 | -2 | -7 | 0 |
| Recycling | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | | -15 | -44 | -51 | -47 | -40 |
| Social costs | *€/ton* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* | *Mln € per year* |
| Landfilling | 45.04 | 609 | 48 | 0 | 0 | 0 |
| Incineration | 111.57 | 0 | 589 | 709 | 589 | 589 |
| Co-incineration | 124.34 | 0 | 66 | 66 | 199 | 0 |
| Recycling | 0 | 0 | 0 | 0 | 0 | 0 |
| *Total* | | 487 | 688 | 758 | 887 | 589 |

# 6. Conclusions

In this concluding chapter, we return to the three central questions that were formulated in Chapter 1:

1. To what extent is the Dutch landfill tax providing the right incentives to the appropriate market parties, and is the waste market functioning sufficiently in order to achieve the policy objectives by means of market incentives?
2. To what extent does the landfill tax cover the external costs of landfilling?
3. Is the landfill tax the least expensive option to minimize landfilling, or are there other ('command-and-control type') instruments that are more cost-effective?

## 6.1 Incentives and functioning of the market

*Has landfilling become a relatively more expensive waste treatment option compared to alternatives such as incineration and recovery/re-use/recycling?*

In The Netherlands, landfilling became more expensive than incineration around the year 2000. Whether or not landfilling is more expensive than recycling is less clear, because there are various types of recycling (such as glass, paper, textiles, etc). Moreover, the choice between recycling and disposing waste is taken at a different level, i.e. the waste generator and not the waste disposer. This is especially true in the case of household waste. Households choose between recycling or disposing waste, while municipalities choose between incinerating and landfilling. For the service sector, individual companies take both decisions.

*Has the landfill tax led to a relative increase in waste supply for incineration and recovery/re-use/recycling, and to a relative decrease in waste supply for landfilling?*

The landfill tax did not have a significant direct impact on the generation of household waste, nor did it affect the choice for household waste disposal options. In the case of household waste generation, the increases in the waste disposal costs, such as the landfill tax, are not internalised in the waste disposal charges in the case of flat fee regimes. In the case of unit-based pricing regimes, the disposal costs are more likely to be internalised in the waste disposal charges. However, to analyse this indirect effect requires an additional analysis of the impact of waste disposal costs on the tariffs of unit-based pricing regimes at the level of municipalities. The presence of a unit-based pricing regime decreases household waste generation. In addition, it decreases the amount of waste landfilled, while it increases the amount of waste incinerated. In the case of disposal options, municipalities (or their contracted collectors of household waste) face the landfill ban on household waste and moreover many municipalities have long-term contracts on waste supply with waste incineration plants.

For the service sector, the level of disposal costs is not affecting the generation of waste, although it affects the waste disposal choice. In particular, higher costs for landfilling and incineration increase the share of recycling. Moreover, if the relative increase of costs of landfilling exceeds the relative increase of costs of incineration, firms from the service sector will landfill less waste and incinerate more. In this sense, the landfill tax can play a crucial role in the decision making of firms from the service sector with respect to disposing waste.

*Has the landfill tax led to a better utilisation of the existing infrastructure for incineration and recovery/re-use/recycling, both in The Netherlands and abroad?*

On the one hand we observe that the capacity of incineration has been fairly constant since 1997. The number of waste incineration plants was constant, and only little fluctuations in capacity were observed. On the other hand, the total amount of waste incinerated increased slightly in the corresponding period. In addition, the relative price of landfilling tariffs and incineration costs has had a positive effect on the amount of service sector waste that has been incinerated. This means that higher costs of landfilling due to higher landfill tax levels will increase the demand of the service sector for the incineration of waste. Due to a constant capacity and a slightly growing incineration of waste, the efficiency of use of waste incineration plants shows a slightly positive trend. In the case of recycling, we observed a moderate growth in the case of household waste, and a strong growth of recycling in the case of the service sector waste. We were not able to link these findings to the efficiency of the use of recycling infrastructure.

*Has the landfill tax led to investments in new capacity for incineration, separation and recovery/re-use/recycling, both in The Netherlands and abroad?*

During the observed period, the landfill tax has not led to new investments in new incineration plants in The Netherlands (though some new plants are presently under construction). However, export of waste for incineration and use as a secondary fuel has increased, so the landfill tax in the Netherlands might have contributed to a more efficient use of foreign waste incineration capacity and/or new foreign incineration capacity. More research into developments of the European waste market is needed to come to a reliable conclusion on this issue. If the landfill tax has led to investments in recycling infrastructure is ambiguous, because different kinds of recycling options require their own infrastructure. In future research, the development of recycling options could be mapped in more detail. The large and increasing export of waste paper to South-East Asia suggests that global transport and recycling industries benefit from the source separation efforts of consumers and companies in The Netherlands (and Europe as a whole).

*What are the conditions that the waste market has to fulfil in order to apply the landfill tax as an effective instrument?*

The analysis suggests a number of important conditions. There should be no restrictions on the availability of incineration capacity (both within The Netherlands and abroad). The limited capacity of incineration plants poses a barrier to the effectiveness of the landfill tax because it limits the options producers have to get rid of their waste. Producers cannot increase incineration and therefore only have the option of recycling more waste (which may be a more expensive option). Allowing export of combustible waste and thus effectively solving the capacity problems increases the effectiveness of the landfill tax, assuming that there is sufficient capacity available in neighbouring countries. Flat fee pricing of waste collection also influences the effectiveness of the landfill tax. Households will not increase their recycling efforts if they do not ‘feel’ the higher cost of landfilling. However, even in municipalities that ask a unit-based price for waste collection the increase in recycling effort is slight. Therefore in this case one can question whether it is necessary to introduce unit-based pricing as the costs of introducing such as system may be higher than the benefits provided by higher recycling efforts. Finally, allowing waste export to be landfilled abroad will restrict the effectiveness of the landfill tax significantly if this effectiveness is measured in terms of 'the reduction in the total amount of waste landfilled in general'. However, if effectiveness is measured in terms of 'the reduction in the amount of waste landfilled in the Netherlands', allowing waste export for landfilling in other countries would turn the landfill tax into a very effective instrument.

*What is the desired or optimal rate of the waste tax in order to make the suppliers of waste choose alternative waste treatment options (incineration and recovery/re-use/recycling)? In particular, should the rate be differentiated according to certain aspects such as type of waste, treatment option and waste supplier?*

At the present landfill tax rate of more than € 80 per tonne it is already attractive for a lot of waste suppliers to turn to alternatives. However, due to other restrictions (such as a lack of incineration capacity, a ban on the export of waste or long-term contracts with landfill operators) they may be forced to landfill their waste anyway. In the absence of such restrictions, higher levels of the landfill tax rate would lead to less landfilling, but at the expense of relatively high social costs. The analysis does not allow us to draw conclusions on the impact of a differentiation of the tax rate by type of waste, treatment option or waste supplier. However, economic theory tells us that any differentiation should only be based upon differences in external costs of the waste and the treatment option, and not on the type of waste supplier.

## 6.2 External costs

*What are the social costs and benefits of the following treatment options for household waste and comparable waste from firms:*

- *Landfilling all waste;*
- *The present situation;*
- *Terminating the landfilling of combustible waste by shifting it to waste incineration;*
- *Terminating the landfilling of combustible waste by shifting it to co-incineration (in power plants);*
- *Terminating the landfilling of combustible waste by optimizing/maximizing recycling.*

Roughly speaking, the net social costs of these five options are estimated at the following (rounded) amounts per year:

- Landfilling all waste: € 500 mln;
- Present situation: € 700 mln;
- Shift from landfilling to incineration: € 750 mln;
- Shift from landfilling to co-incineration: € 900 mln;
- Optimizing/maximizing recycling: € 600 mln.

Given the large uncertainties and data gaps, these amounts should be regarded as mere tentative estimates.

*To what extent can the current level of the Dutch landfill tax rate be regarded as an internalisation of the environmental costs of landfilling?*

The calculations made strongly suggest that the current landfill tax rate of almost € 85 per tonne is at least equal to, but probably substantially higher than the marginal external costs of landfilling. The 'best' estimate of the external costs is about € 9 per tonne, and even the 'high' estimate (€ 75 per tonne) is lower than the current tax rate.

*From a social cost-benefit perspective, what is the optimum way of waste treatment for household waste and comparable waste from firms?*

A definitive answer to this question cannot be given, due to lacking and sometimes contradictory information, especially regarding co-incineration and various recycling options. Nevertheless, it is clear that the net social costs of landfilling are probably much lower than those of (co-)incineration. Some combination of landfilling (with methane recovery) and recycling might well be the strategy implying minimum net social costs. Obviously, such a strategy would be dependent on the availability of sufficient space for (new) landfills.

## 6.3 Cost effectiveness of alternative instruments

*What are the financial consequences (in the short and the long term) of landfill taxation, landfill bans and legal obligations (for landfills and incineration plants) to accept waste, both for the waste suppliers and the waste treatment companies?*

In economic terms, a landfill ban is similar to a prohibitively high landfill tax rate (about € 750 per tonne). Our analysis shows that the social costs of reducing the amount of landfilled waste to zero would be very high. Reducing the amount of landfilled waste (from the service sector) by about 90% costs about 600 million euro in production loss, while reducing it by 100% costs about 1,300 million euro in production loss. A similar calculation could not be made for the case of legal obligations to accept waste, but it is obvious that such an obligation would also imply welfare losses (assuming that it would force waste treatment companies to undertake unprofitable operations).

*Which instrument will (in the short and the long term) lead to the lowest costs of waste treatment for the waste suppliers?*

A well-designed system of environmental taxes will minimize the total social costs of waste treatment, provided that there are no market distortions. This means that waste suppliers should have the opportunity to choose between alternative treatment options (domestically or abroad) and that the tax rates reflect the external costs of the treatment option (landfilling as well as incineration)

In the current Dutch situation, this would probably mean a reduction of the landfill tax rate and an increase in the waste tax rate for incineration (which at present has a zero rate). Both rates should be around € 10 per tonne of waste. However, it is clear that a waste tax of this magnitude would reduce the incentive to divert waste from landfilling to recycling and incineration substantially. Alternatively, tradable landfill permits could be considered. This instrument, which combines the efficiency advantages of a tax with the certainty of a cap on the total amount of landfilled waste, deserves further investigation.

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# Appendix I. Calculating $CH_4$ emissions from landfills

To assess the volume of $CH_4$ emissions from landfill waste in the Netherlands we use the methodology and assumptions of RIVM (Spakman *et al.*, 2003) that are used in the official calculations of Dutch GHG emissions. Landfill wastes emit $CH_4$ gas over a period of time. The annual emissions are dependent upon the composition of the waste (the biodegradable organic fraction), a decay factor, the rate of landfill gas recovery, and a number of technical coefficients. In formula:

$$Z_t = f \times k \times P \times s \times \left[1 - a(t)\right] \times b \times {}^{16}\!/_{12} \times e^{-kt} \quad \text{(I.1)}$$

Where:

$Z_t$ = $CH_4$ emissions in year t (in ton/ton waste);
$f$ = the fraction of biodegradable waste that is actually broken down ($f$ = 0.58 ton C/ton waste);
$k$ = decomposition (decay) constant ($k$ = 0.0693) ;
$P$ = concentration of biodegradable waste ($P$ = 0.120 ton C/ ton waste);
$s$ = share of carbon emitted in the form of $CH_4$ ($s$ = 0.6);
$a(t)$ = landfill gas recovery rate in year $t$;
$b$ = fraction of non-recovered $CH_4$ that is actually emitted to the atmosphere ($b$ = 0.9);
16/12 = conversion factor from mass C to mass $CH_4$;
$t$ = time since disposal (in years).

Because the formula (A1) contains several time-independent coefficients, it can be simplified to:

$$Z_t = 0.003473 \times \left[1 - a(t)\right] \times e^{-0.0693t} \quad \text{(I.2)}$$

Total $CH_4$ emissions per ton of waste over an entire period t = 0...T can be calculated as the sum of the annual emissions of $CH_4$:

$$Z = \sum_{t=0}^{T} 0.003473 * \left[1 - a(t)\right] * e^{-0.0693*t} \quad \text{(I.3)}$$

In our calculations we have assumed that $T$ = 30.

# Appendix II. Spatial distribution of landfills and incinerators in the Netherlands (in Dutch)

## Bepaling geografische positie en omtrek van de stortplaatsen

Voor een nauwkeurige bepaling van woningdichtheden rond stortplaatsen en AVI's zijn de geografische locatiegegevens van deze eenheden nodig en de recente ruimtelijke verdeling van huizen rondom deze locaties.

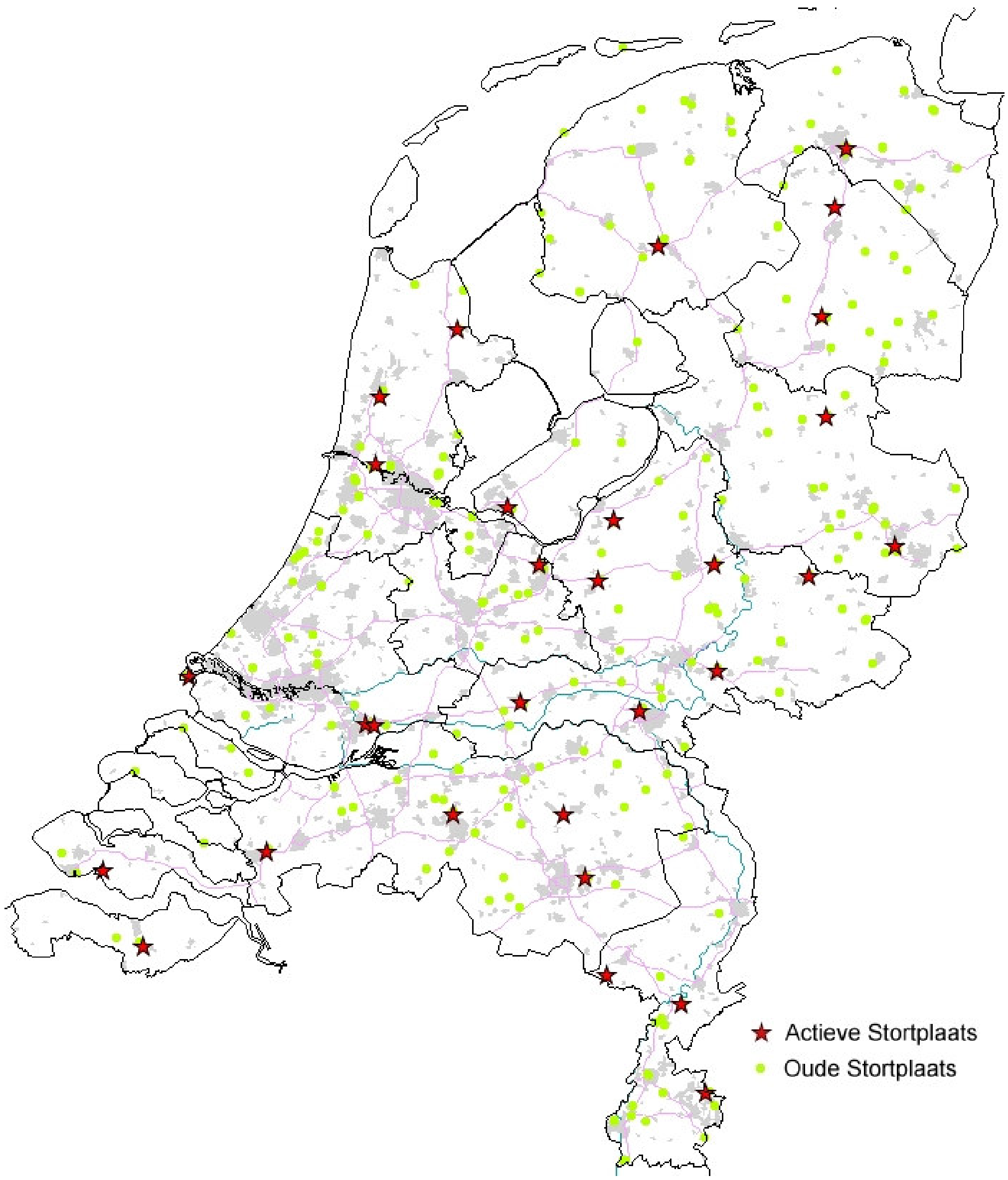
De geografische locatiegegevens kunnen in eerste instantie afgeleid worden van de beschikbare adresgegevens. Een adresbestand met geografische coördinaten (zoals de bestanden van het kadaster Adres Coördinaten Nederland (ACN) en/of het Perceel Adres Plaats coördinaten (PAP)) is bij het SPINlab niet beschikbaar. Andere alternatieven zoals een online dienst om de xy-coordinaten bij adressenbestanden te verkrijgen, blijken helaas niet te werken (zie onderdeel geo-coderen op http://www.thebitfactory.nl/).De locaties van straten en/of postcodes moeten daarom op de kaart gevonden worden met behulp van de Microsoft MapPoint applicatie of met online kaartdiensten als de Nationale Telefoongids of Map24. Met behulp van een of meerdere van deze applicaties konden de meeste adressen gevonden worden. Maar in sommige gevallen kon de exacte positie van het adresnummer in de betreffende straat niet met zekerheid vastgesteld worden. De bron voor de exacte adreslocatie en de veronderstelde nauwkeurigheid is in de spreadsheet met stortplaatsen en AVI's voor de verschillende in een aparte kolom aangegeven. De geografische xy coordinaten (lat-long) zijn vervolgens omgezet naar de xy coordinaten in de stereografische projectie van het Nederlandse Rijksdriehoekstelsel. Hiervoor is een ArcView extentie RD-conversions gebruikt, maar dit kan ook gebeuren aan de hand van verschillende online coordinatecalculators.

De xy-coordinaten van de hierboven bepaalde locaties zijn geplot in een digitale kaart van Nederland (zie Figuur II.1) en als GIS laag toegevoegd aan de Bodemstatistiekkaart 2000.

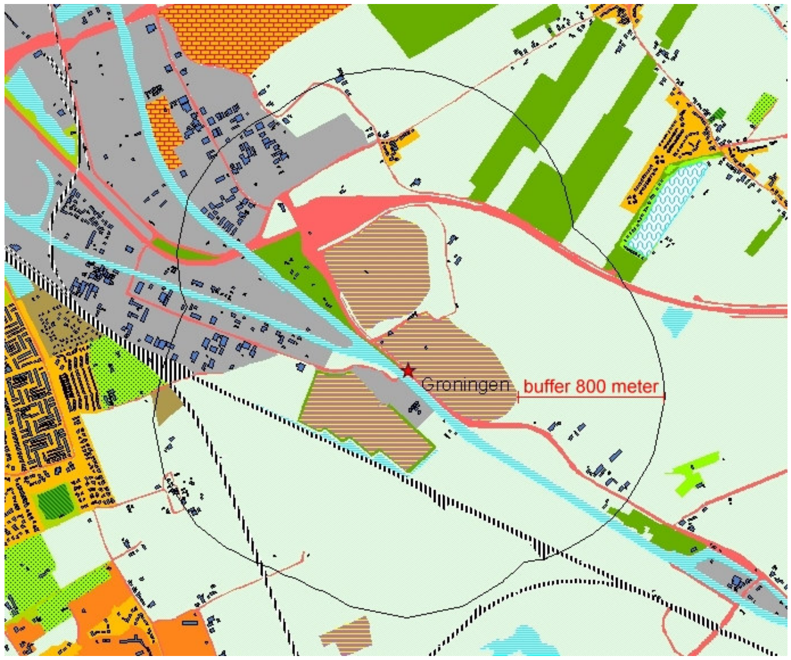
Aangezien op de bodemstatistiekkaart stortplaatsen als aparte categorie staan aangegeven kunnen de polygonen van de actieve stortplaatsen geselecteerd worden op basis van de gemaakte overlay van het stortplaats puntenbestand en de bodemstatistiekkaart. De geselecteerde stortplaatsen worden vervolgens als een aparte GIS-laag opgeslagen voor gebruik in verdere analyses.

Uit deze overlay bleek dat de methode voor het bepalen van de xy-coordinaten redelijk gewerkt heeft, maar laat tevens de beperkingen van deze methode zien. Het bezoekadres van de stortplaats heeft normaal betrekking op het kantoor bij de ingang van de stortplaats, wat op enige afstand van de stortplaats gelegen kan zijn. Met andere woorden, de aldus bepaalde xy coördinaten verwijzen niet naar het middelpunt van de stortplaats zelf. Verder, omdat dit een puntlocatie betreft geeft het ook geen maat voor de geografische omvang van de locatie. Daardoor is het uitvoeren van een bufferoperatie rondom de stortplaats om een invloedsgebied te kunnen aangeven, van beperkte waarde omdat geen rekening wordt gehouden met de vorm en de grootte van de stortplaats. Hiervoor is het beter de buffer vanaf de randen van de stortplaats te tekenen, wat kan met behulp van de geselecteerde stortplaatsen uit de bodemstatistiekkaart. Figuur II.2 laat als voorbeeld de stortplaats 'De Stainkoeln' in Groningen zien. De rode ster geeft de xy-positie aan die is gevonden mbv de adreslocatiemethode. De-

ze
locatie komt in dit geval goed overeen met het middelpunt van de stortplaats die uit 3 of 4 verschillende delen bestaat (de vlakken met de horizontale bruingestreepte arcering). Rondom 3 van de 4 vlakken is een bufferzone van 800 meter gedefinieerd.




*Figuur II.1: Actieve en oude stortplaatsen in Nederland.*

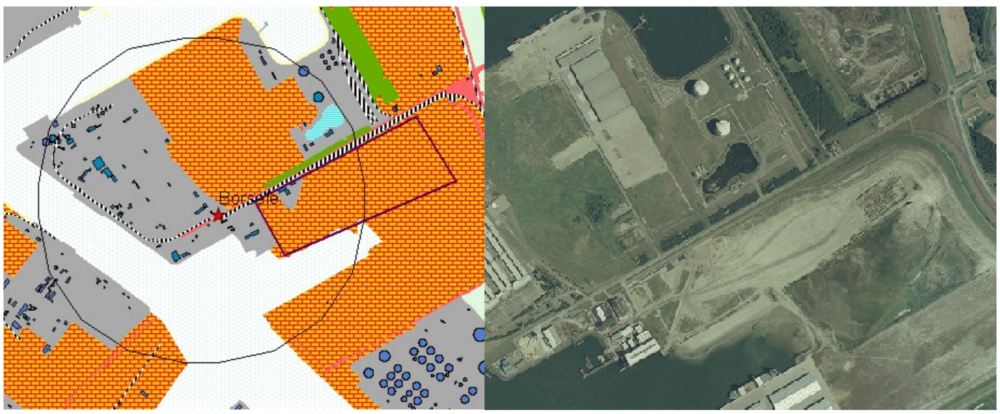



*Figuur II.2 Stortplaats De Stainkoeln' in Groningen op Bodemstatistiekkaart met 800 meter buffer.*

Drie actieve stortplaatsen bleken niet voor te komen op de bodemstatistiekkaart, waarschijnlijk omdat deze pas na 2000 (het data-inwinningsjaar van de bodemstatistiekkaart) in gebruik zijn genomen. De vermoedelijke locatie en omtrek van de stortplaatsen zijn in dit geval bepaald met behulp van ondersteunende luchtfoto's uit het jaar 2003 die online geraadpleegd kunnen worden vanaf de website 'Beeldportal': http://www.beeldportal.nl/

Figuur II.3a, II.3b en II.3c laten van de drie stortplaatsen de luchtfoto's zien en de ingetekende vermoedelijke locaties op de bodemstatistiekkaart. In Figuur II.3a is bijvoorbeeld te zien dat in 2000 de stortplaatslocatie nog in gebruik was voor delfstoffenwinning volgens de definitie van de bodemstatistiekkaart (het lichtpaarse gedeelte betreft water met delfstofwinningsfunctie). Ook is op deze kaart goed het effect te zien van de grootte van de bufferzone. In dit geval valt het dorp ten westen van de stortplaats net buiten de bufferzone van 800 meter. In Figuur II.3b is de locatie van de stortplaats gedefinieerd als bouwterrein volgens de classificatie van de bodemstatistiekkaart. Op basis van de luchtfoto is niet met zekerheid te zeggen waar de stortplaats zich bevindt. Maar gezien de zichtbare op- en afrij routes op het zandige gedeelte onder in de foto lijkt dit de meest waarschijnlijke locatie van de stortplaats, die dan ook als zodanig op de bodemstatistiekkaart is ingetekend. Overigens is de aangegeven buffer hier nog getrokken rondom de puntadreslocatie en is dus onjuist.

*Figuur II.3a Stortplaats Meisner in Assen.*




*Figuur II.3b Stortplaats Noord en Midden Zeeland in Borsele.*

In Figuur II.3c zijn de stortplaatsen in Dordrecht weergegeven. Goed te zien is dat de adres puntlocatie van de Stortplaats Derde Merwedehaven hier minder goed overeenkomt met de werkelijke locatie van de stortplaats. Meer problematisch is de tweede stortplaats Crayestein-West die niet is aangegeven op de Bodemstatistiekkaart. Ook de luchtfoto geeft hier geen duidelijk uitsluitsel. De meest waarschijnlijke locatie lijkt het braakliggende terrein direct ten oosten van de snelweg op de luchtfoto.

In bijlage 1 zijn alle actieve stortplaatsen weergegeven met XY-coordinaten en de bron voor deze coördinaten en eventuele onzekerheden.

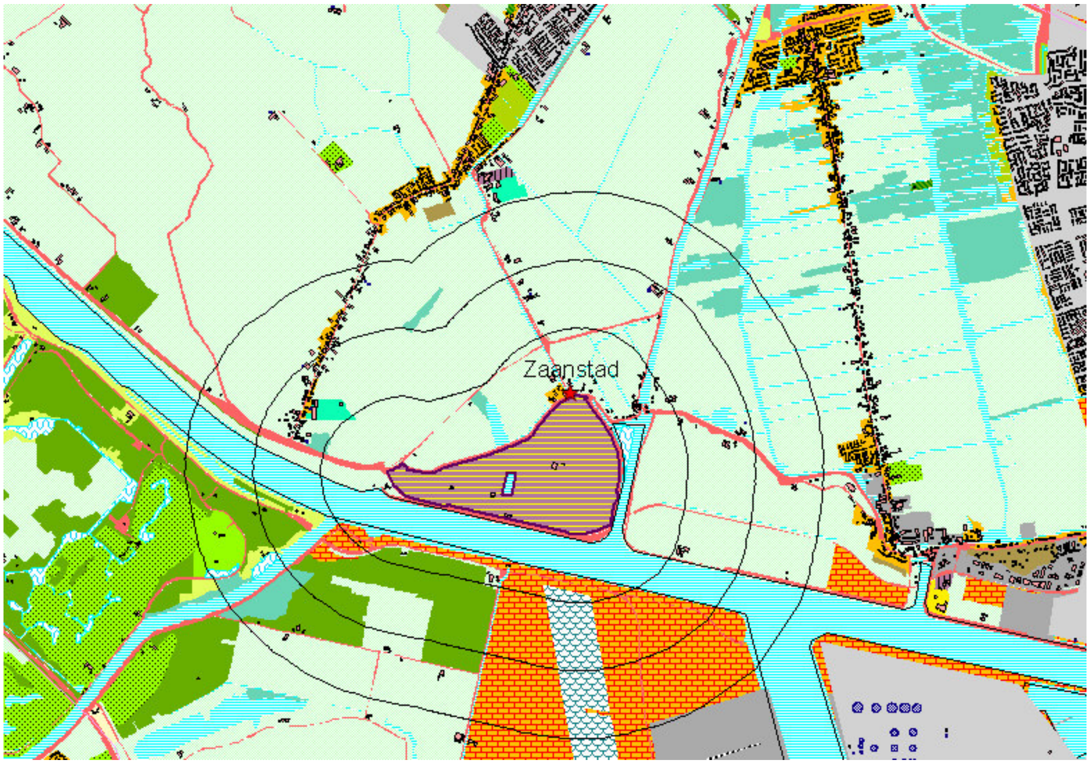
## Bepaling woningdichtheid rond actieve stortplaatsen

Woningdichtheid wordt gebaseerd op het aantal gebouwen aangegeven op de topografische kaart van Nederland (schaal 1:10.000) van 2003. Onder gebouwen wordt door de TDN verstaan de categorieën 100 (gebouw/huis), 103 (hoogbouw) en 108 (opslagtank). In deze analyse worden alleen de categorieën 100 en 103 meegenomen in de analyse en worden alleen gebouwen van minimaal 30 m$^2$ in de analyse betrokken, om zo min mogelijk bijgebouwen (schuren, garages e.d.) als huizen te laten meetellen.


*Figuur II.3c Stortplaats Crayestein-West (links) en Derde Merwedehaven (rechts) in Dordrecht.*

De GIS laag met bebouwing is een aparte laag die is opgesplitst per kaartvlak. Ter voorbereiding van de analyse zijn alle kaartvlakken bepaald waarin de actieve stortplaatsen gelegen zijn, inclusief de daarom heen te bepalen bufferzone. Vervolgens zijn van de GIS lagen met bebouwing alleen deze kaartvlakken geselecteerd en in 1 kaartlaag opgenomen met behulp van een ‘merge’ operatie. Hiermee is een handzame laag gecreëerd voor de uit te voeren analyses qua grootte bestandsformaat.

Vervolgens is een bufferoperatie uitgevoerd op de laag met actieve stortplaatsen. In eerste instantie zijn buffers gecreëerd van 500, 1000 en 1500 meter, zie figuur 4.




*Figuur II.4. Buffers van 500, 1000 en 1500 meter rond stortplaats Zaanstad.*

Vervolgens zijn selecties uitgevoerd van alle gebouwen die binnen een van de bufferzones vallen (respectievelijk 500, 1000 en 1500 meter). In ArcView betreft dit een zogenaamde ‘Select by theme’ operatie waarbij de GIS laag met gebouwen als actief thema wordt gekozen en een van de bufferlagen als laag waarbinnen de selectie moet worden uitgevoerd, dat wil zeggen dat alleen polygonen geselecteerd worden die compleet binnen de bufferzones vallen. Van deze selectie gebouwen zijn vervolgens alleen de gebouwen groter dan 30 $m^2$ geselecteerd in een query operatie (select from set) op de attribuuttabel van de laag met gebouwen. Deze set met gebouwen is vervolgens als een aparte laag weggeschreven waarvan de statistieken bepaald kunnen worden. In dit geval betreft dit alleen het totaal aantal polygonen. Omdat in het aantal huizen nu ook de huizen of gebouwen zitten die op het terrein van de vuilstort zelf gelokaliseerd zijn, moet dit aantal huizen apart bepaald worden en van het aantal huizen binnen de bufferzone afgetrokken worden. Het aantal gebouwen/huizen groter dan 30 $m^2$ op de vuilstortlocatie zelf bedraagt 163. Door de oppervlakte te berekenen van de bufferzones kan vervolgens de polygonendichtheid berekend worden (zie Tabel II.1). De oppervlakte van de bufferzones wordt berekend door de oppervlaktes van alle polygonen per bufferlaag bij elkaar op te tellen (dit gebeurt met de statistics functie in de attribuuttabel). Aangezien de bufferlagen anders gedefinieerd zijn dan de polygoonlagen waar ze op gebaseerd zijn, moet eerst een conversie worden uitgevoerd om de oppervlakten van de afzonderlijke polygonen te berekenen. Dit laatste gebeurt met behulp van de XTools extensie in ArcView (Convert Multipart Shapes to Single Part Shapes). Bij de berekening van de bufferoppervlakte wordt steeds de oppervlakte van de stortplaats van de totale bufferoppervlakte afgetrokken.

*Tabel II.1 Oppervlaktes invloedsgebieden, aantal polygonen en polygonendichtheid.*

| Invloedsgebied rond stortplaats | Oppervlakte in km$^2$ | Aantal polygonen | Polygonendichtheid (aantal/km$^2$) |
|---|---|---|---|
| Stortplaats zelf | 13,84 | 163 | Nvt |
| 500 meter, excl. stortpl | 72,28 | 2516 | 34,8 |
| 1000 meter, excl. stortpl | 187,98 | 10280 | 54,7 |
| 1500 meter, excl. stortpl | 347,47 | 22417 | 64,5 |

Uiteraard kan er ook apart gekeken worden naar de dichtheden in gebieden op bepaalde afstand van de stortplaatsen, bijvoorbeeld van 1000 tot 1500 meter. Hiervoor hoeft alleen de oppervlakte van de 1000 meter buffer van de 1500 meter buffer afgetrokken te worden en evenzo het aantal huizen in de 1000 meter buffer van de 1500 meter buffer, waarna beide getallen door elkaar gedeeld kunnen worden om de dichtheid te berekenen.

Laatste stap in deze analyse is een correctie van het aantal polygonen. Dit is nodig omdat er niet van uit kan worden gegaan dat het aantal getelde polygonen in de huizenkaart ook het werkelijke aantal huizen betreft. Dit komt doordat aaneengeschakelde huizen (rijtjeshuizen, 2 onder 1 kap, appartementcomplexen) als 1 polygoon weergegeven worden.

Om het werkelijke aantal huizen te benaderen zijn we van het volgende uitgegaan. De meeste niet gestapelde huizen in Nederland hebben een bebouwde oppervlakte tussen de 100 en 200 m$^2$ (eigen schatting, mede op grond van metingen aan huizenbestand topografische kaart). Er zijn uiteraard grotere huizen, maar de meeste polygonen > 200 m$^2$ betreffen rijtjeshuizen, 2 onder 1 kap e.d. Ervan uitgaande dat de meeste polygonen > 200 m$^2$ uit meerdere huizen bestaan kan daarom een selectie gemaakt worden uit het bestand van alle polygonen > 200 m$^2$ en kan dit aantal gedeeld worden door een gemiddelde huisoppervlakte van bijv. 150 m$^2$. Het aantal huizen wat hier uit komt wordt vervolgens opgeteld bij alle polygonen (losstaande huizen) tussen de 30 en 200 m$^2$.

Allereerst moeten de polygonen van symboolklasse 108/109 (opslagtanks) uit het bestand verwijderd worden. Van de 22.094 overblijvende polygonen van het 1.500 meter invloedsgebied zijn er 8.501 > 200 m$^2$. Deze hebben een gezamenlijke oppervlakte van 4.506.914 m2 / 150 = 30.046 huizen. Als we hier het aantal van 13.593 huizen < 200 m$^2$ bij optellen komen we uit op 43.639 huizen. De huizendichtheid komt dan uit op 126 huizen per km$^2$.

*Tabel II.2 Oppervlaktes invloedsgebieden, gecorrigeerd aantal huizen en huizendichtheid.*

| Invloedsgebied rond stortplaats | Oppervlakte in km$^2$ | Aantal huizen | Huizendichtheid (aantal/km$^2$) |
|---|---|---|---|
| Stortplaats zelf | 13,84 | 163 | nvt |
| 500 meter, excl. stortpl | 72,28 | 5390 | 75 |
| 1000 meter, excl. stortpl | 187,98 | 20355 | 108 |
| 1500 meter, excl. stortpl | 347,47 | 43639 | 126 |

Deze aantallen worden groter naarmate de invloedszone uitgebreid wordt en lopen op richting de gemiddelde dichtheid van 200 woningen per km$^2$ in 2003 in Nederland (CBS StatLine, 2005).

Ter controle van de berekeningswijze is ook een analyse uitgevoerd op basis van het woonareaal in het bodemstatistiek 2000 bestand. Dit levert de volgende resultaten op.

*Table II.3 Oppervlaktes invloedsgebieden, woonareaal en afgeleide huizendichtheid.*

| Invloedsgebied rond stortplaats | Oppervlakte in $km^2$ | Woonareaal | Aantal huizen | Huizendichtheid (aantal/$km^2$)[1] |
|---|---|---|---|---|
| Stortplaats zelf | 13,84 | 163 | | Nvt |
| 500 meter, excl. stortpl | 72,28 | 645869 | 4306 | 75 |
| 1000 meter, excl. stortpl | 187,98 | 5679898 | 37866 | 201 |
| 1500 meter, excl. stortpl | 347,47 | 32409241 | 216062 | 621 |

[1] Uitgaande van een oppervlakte van ca. 150 $m^2$ per huis.

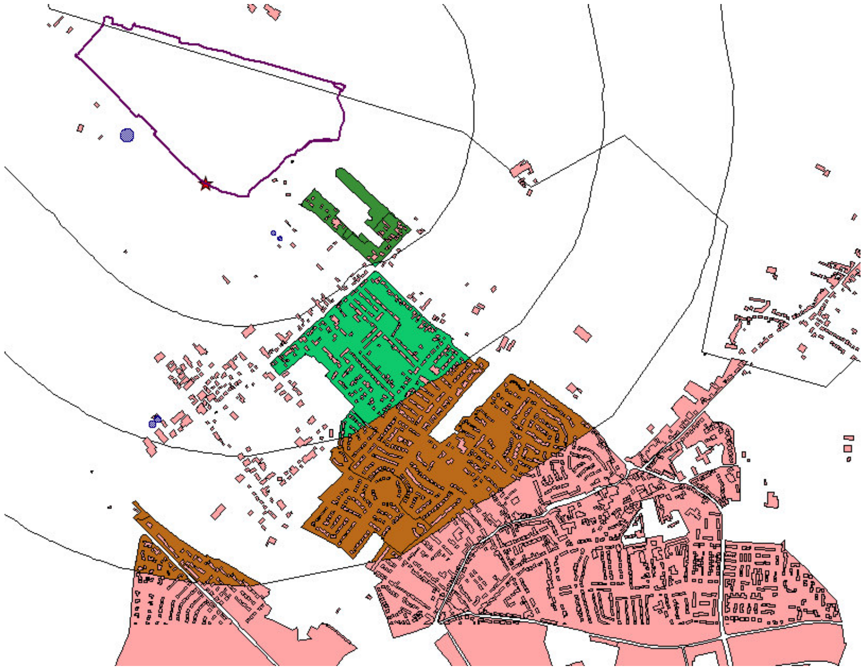
Dit is duidelijk geen representatief beeld, wat een gevolg is van de afbakening van het woonareaal en de verschillen in huizendichtheid die voorkomen op dit woonareaal (zie Figuur II.5). Een oppervlakte van 150$m^2$ per huis is duidelijk veel te klein, maar door de verschillen in bebouwingsdichtheid is het kiezen van een goede oppervlaktemaat niet mogelijk. Om dit te berekenen moet de zogenaamde 'Ground Space Index' (GSI) bekend zijn. De GSI geeft de verhouding tussen bebouwde en onbebouwde ruimte weer (Gemeente Amsterdam, 2002). Ook wordt duidelijk uit de figuur dat een deel van de huizen niet als woonareaal zijn geclassificeerd.

## Uitsplitsing resultaten naar stortplaats

Door de attribuuttabellen van de GIS lagen met de polygonen en buffers rond de stortplaatsen uit te breiden met een kolom met de stortplaatsnamen kan per stortplaats de totale oppervlakte berekend worden. Door vervolgens een 'intersect' operatie[58] uit te voeren tussen de bufferlagen en de lagen met huizen die binnen een bepaalde buffer vallen wordt aan elk huis de naam van de bijbehorende stortplaats gekoppeld en kunnen per stortplaats en per bufferzone het aantal en dichtheid huizen berekend worden.

Om bijvoorbeeld het aantal huizen in een 500 meter zone rond stortplaats De Stainkoeln' te Groningen te berekenen, kunnen alle records in de tabel met STRTPLAATS naam 'De Staink Groningen' bij elkaar opgeteld worden (in dit geval precies 100 polygonen). Hier moet vervolgens het aantal gebouwen gelegen op de stortplaats zelf afgetrokken worden; dit betreft 3 huizen/gebouwen. Ook moeten de records van symboolklasse 108 (opslagtanks) uit het bestand verwijderd worden. Dus het totale aantal polygonen rond stortplaats De Staink te Groningen is 97. Om het exacte aantal huizen te berekenen, moeten we in principe dezelfde correctie als hierboven uitvoeren, dat wil zeggen dat alle polygonen > 200 $m^2$ bij elkaar opgeteld worden en gedeeld worden door 150 om het aantal huizen > 200 $m^2$ te berekenen. Hier wordt het aantal huizen tussen de 30 en 200 $m^2$ bij opgeteld. Omdat dit voor elke stortplaats afzonderlijk te berekenen erg bewerkelijk is, kan volstaan worden met een eenvoudiger correctie. Namelijk het delen van de totale oppervlakte huizen door een gemiddelde huisoppervlakte van 150 $m^2$.

[58] Hiervoor is de X-Tools extentie gebruikt: Intersect Themes



*Figuur II.5 Woonareaal in bufferzones rond stortplaats Landgraaf.*

Tabel 4 bevat een selectie uit een van de databestanden.

*Tabel II.4 Selectie uit bestand intersecthuizenbuffer500.dbf.*

| SYMBOL | BUFFERDIS | STRTPLAATS | AREA | PERIMETER | ACRES | HECTARES |
|---|---|---|---|---|---|---|
| 100 | 500.0000 | De Staink Groningen | 218.533 | 83.491 | 0.054 | 0.022 |
| 100 | 500.0000 | De Staink Groningen | 318.968 | 77.197 | 0.079 | 0.032 |
| 108 | 500.0000 | Ecopark Skarsterl | 157.292 | 44.826 | 0.039 | 0.016 |
| 100 | 500.0000 | Ecopark Skarsterl | 356.901 | 86.454 | 0.088 | 0.036 |

Dit brengt het aantal huizen rond deze stortplaats op ca. 151. Voor het berekenen van de dichtheid geldt het zelfde, hiervoor moet de oppervlakte van de stortplaats afgetrokken worden (NB: de stortplaats bestaat soms uit meerdere polygonen die bij elkaar opgeteld moeten worden) van de oppervlakte van de 500 meter buffer (die ook de stortplaats zelf bevat).

De oppervlakte van de 500 meter buffer bedraagt in dit geval 3,73 – 0,67 = 3,06 km$^2$, dus een huizendichtheid van 49 huizen per vierkante kilometer. Bij het bepalen van de oppervlakte van buffers is van belang dat sommige buffers van 1 stortplaats uit meerdere delen bestaan, bijvoorbeeld voor stortplaats Smink Amersfoort (bij grotere buffers komen deze bufferdelen weer samen). Bij de twee stortplaatsen in Dordrecht gebeurt het omgekeerde. Deze twee verschillende stortplaatsen liggen zo dicht bij elkaar dat de buffers van 1000 en 1500 meter elkaar raken en daarom als 1 buffer voor een gecombineerde stortplaats voorkomen.

### Beperkingen in de analyse

- Geen onderscheid is gemaakt in het type stortplaats, in verband met ander type overlast, stank, lawaai, verontreinigingsrisico, etc.;
- Brondata van Bodemstatistiekkaart zijn uit 2000. Recente stortplaatsen zijn dus niet aangegeven en zijn derhalve zelf bepaald aan de hand van adresgegevens en beschikbare luchtfoto's. Dit betreft 3 stortplaatsen:
    - Stortplaats Crayestein-West te Dordrecht;
    - Stortplaats Meisner te Assen;
    - Stortplaats Noord en Midden Zeeland in Borsele.
- Brondata voor huizen en gebouwen op de topografische kaart kunnen van verschillende ouderdom zijn, afhankelijk van update-frequentie TDN van de verschillende kaartbladen in Nederland. Hoewel de gegevens nooit ouder dan 10 jaar zijn, kan het bijvoorbeeld voorkomen dat recente nieuwbouwwijken niet aangegeven staan op de kaart. Een vrij eenvoudige manier om dit te controleren is om de huizen en gebouwen te vergelijken met luchtfoto's van 2003 op de website 'Beeldportal': http://www.beeldportal.nl/.

### Verbrandingsinstallaties

In deze uitwerking zijn alleen de onderdelen van het analyseproces die verschillen van het proces voor de stortplaatsen, vermeld. Voor de analyse zijn alleen de kaartbladen van de topografische kaart geselecteerd en samengevoegd van het gebied rondom de AVI's.

Buffers rond AVI's:

- 1000 meter (contiguous)
- 2000 meter
- 3000 meter
- 4000 meter
- 5000 meter

De twee vlak bij elkaar gelegen AVI's in Dordrecht resulteren in een gezamenlijke bufferzone, daar we gewerkt hebben met 'contiguous' buffers. Na het bufferen is een 'multi part shapes' naar 'single part shapes' operatie uitgevoerd om de verschillende bufferzones in de attribuuttabel van elkaar te kunnen onderscheiden en hieraan de namen van de AVI's te kunnen koppelen.

Van de vlakkenkaart is een selectie gemaakt van de vlakken met de volgende symboolnummers (zie figuur 5):

- 101 = bebouwd gebied/huizenblok
- 102 = groot gebouw
- 103 = hoogbouw

Daarna zijn alle polygonen kleiner dan 30 m$^2$ verwijderd en zijn alleen die polygonen geselecteerd die binnen de begrenzingen van één van de volgende landgebruikklassen op de bodemstatistiekkaart vallen:

- 20 = woongebied
- 40 = parken en plantsoenen (sommige losstaande huizen zijn in deze gebieden gelegen)

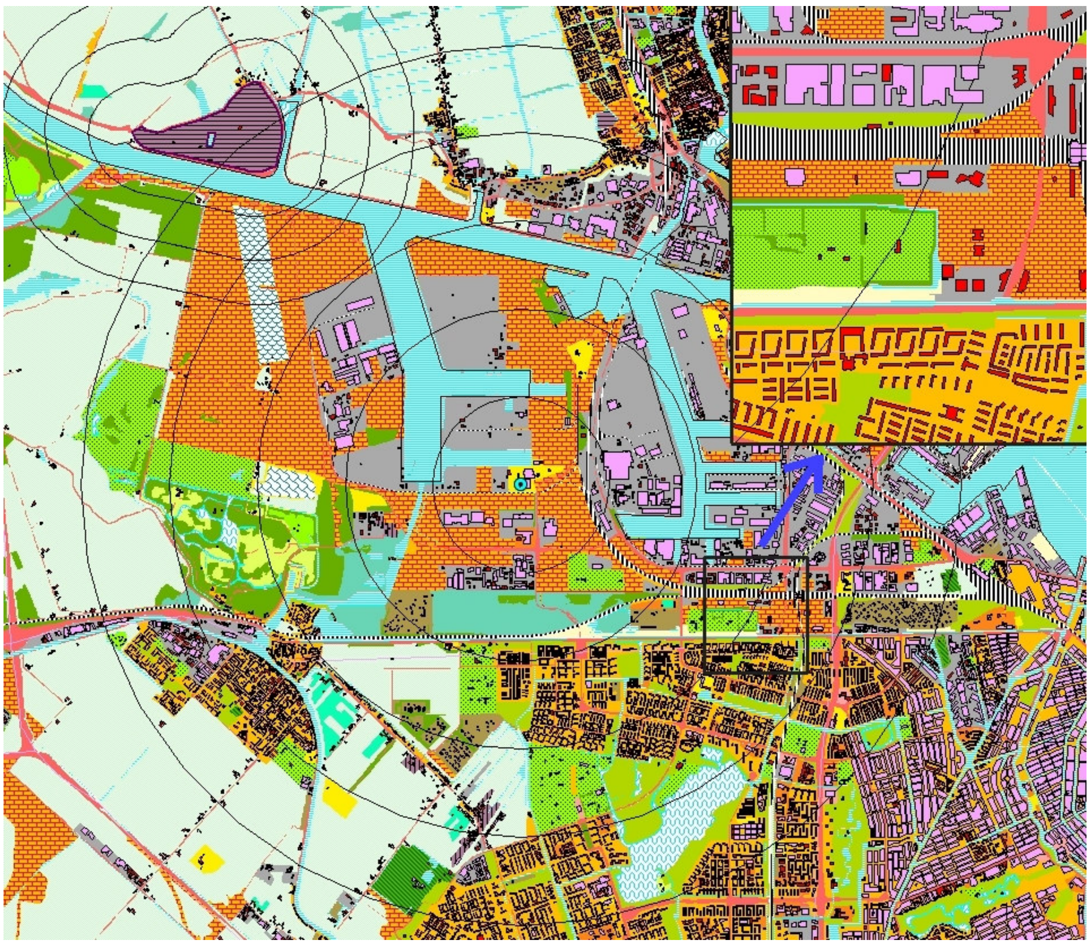
- 60 = bos
- 61 = droog natuurlijk terrein
- 62 = nat natuurlijk terrein

Polygonen die dus bijvoorbeeld binnen de grenzen van industriegebieden, sociale of openbare voorzieningen of die op het terrein van volkstuincomplexen zijn gelegen, zijn hiermee dus verwijderd.

Van de huizenkaart zijn de volgende polygonen geselecteerd:

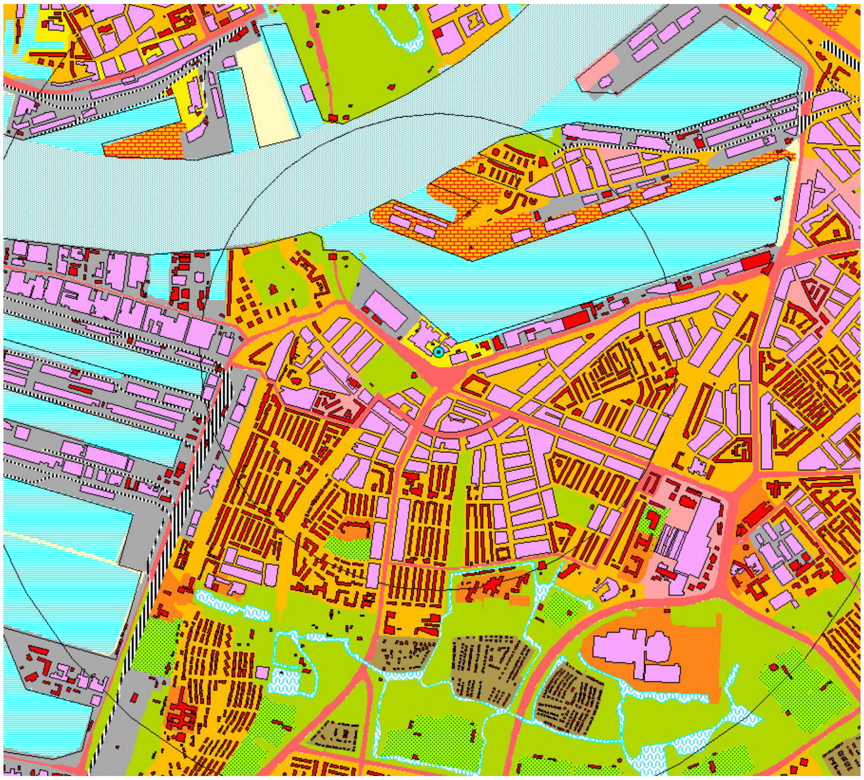
- 100 = gebouw/huis
- 103 = hoogbouw

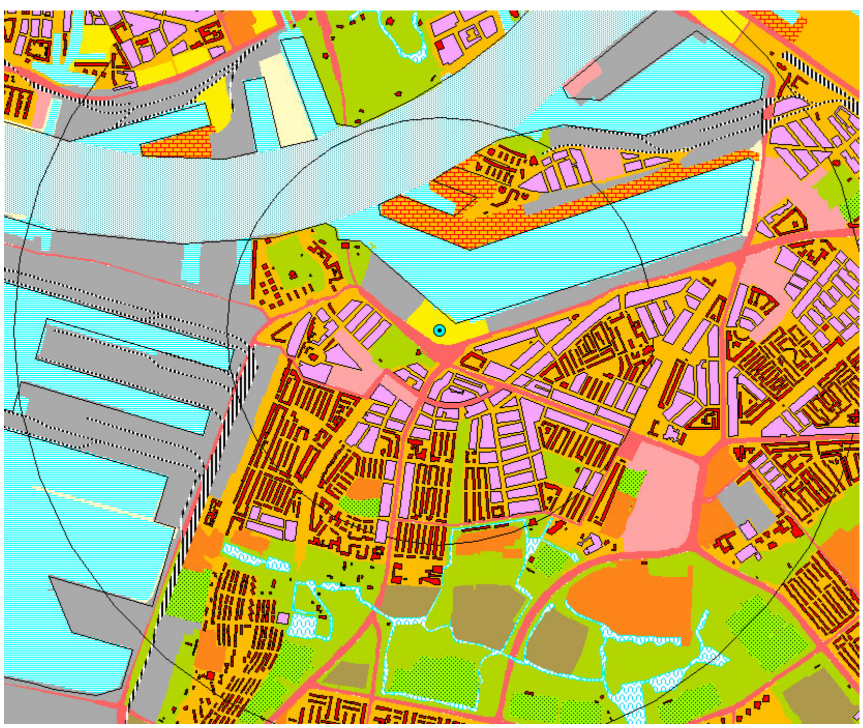
Uit deze selectie zijn de huizen van > 30 $m^2$ geselecteerd die binnen bovengenoemde klassen van de bodemstatistiekkaart vallen.



*Figuur 2.6 Buffers van 1 t/m 5 kilometer rondom het Afval Energie Bedrijf van Amsterdam. Op de ondergrond van de bodemstatistiekkaart zijn de lagen huizen (rode polygonen) en gebouwen (roze polygonen) gelegd, zie ook het detail rechtsbovenin. Ondermeer de roze polygonen die in de grijze industriegebieden van de bodemstatistiekkaart vallen, zullen later worden verwijderd.*

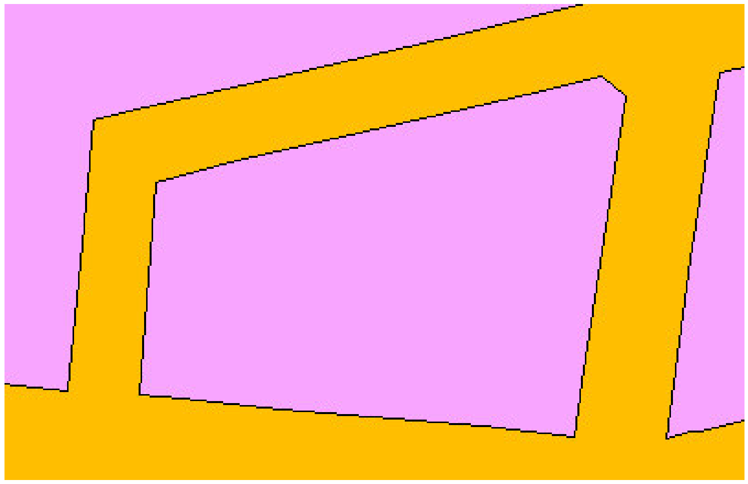
In figuren 6a en 6b, het gebied rond de AVR in Rotterdam aan de Brielseweg, is goed het effect van de selecties te zien op het huizenbestand rondom de AVI's.



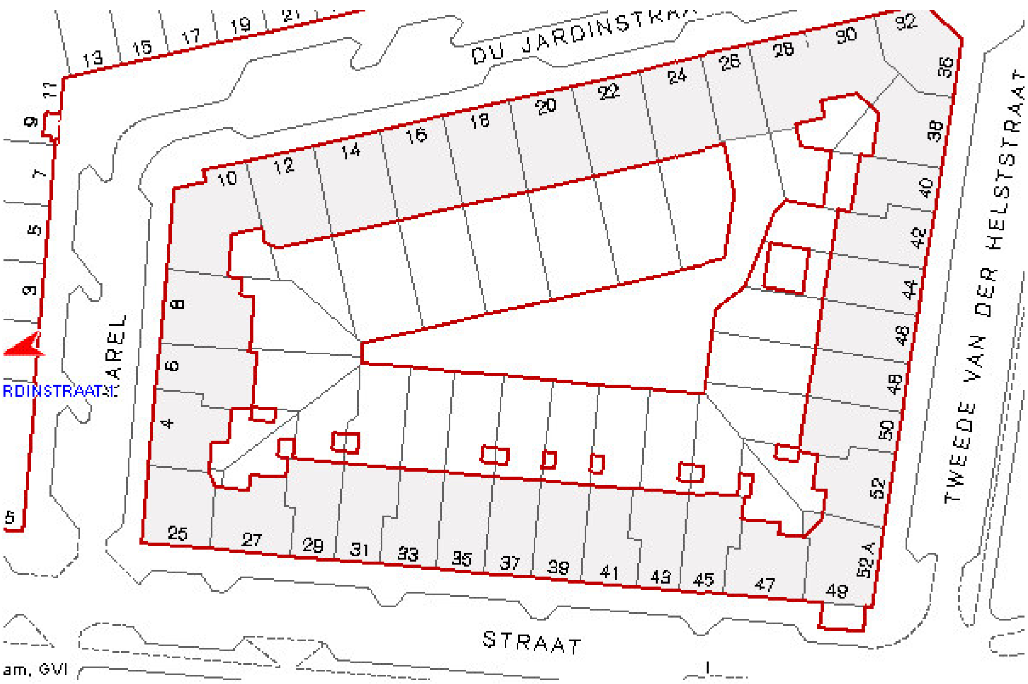


*Figuur II.7a en II.7b. Huizen en huizenblokken vóór en ná selectie van woongebieden.*

Aangezien de huizenblokkenkaart alleen maar de buitenbegrenzingen van de huizenblokken aangeeft, kan hier niet zomaar door een gemiddelde huizenoppervlakte (en het aantal bouwlagen) gedeeld worden om het aantal afzonderlijke huizen te bepalen. Om in te kunnen schatten uit hoeveel huizen een huizenblok bestaat is hieronder een huizenblok uit de door ons afgeleide huizenblokkenkaart (zie figuur 7a) vergeleken met een zelfde huizenblok op de grootschalige basiskaart Nederland (zie figuur 7b), in dit geval een huizenblok betreffende in Amsterdam Oud-Zuid (via website Amsterdam.nl / stadsplattegrond).

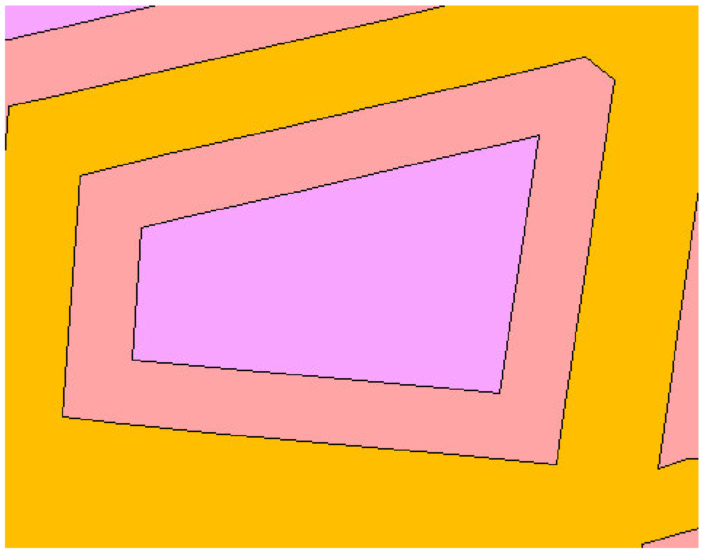


*Figuur II.8a Huizenblok in Amsterdam Oud Zuid op huizenblokkenkaart.*




*Figuur II.8b Huizenblok in Amsterdam Oud Zuid op GBKN*

Deze kaart laat zien, dat een groot deel van de oppervlakte van dit blok wordt ingenomen door de binnenruimte (tuinen + opslagruimte). Aan het aantal huisnummers is te zien dat het hier gaat om 38 verschillende panden. Aangezien het hier panden betreft met 4 bouwlagen (het meest gebruikelijke aantal woonlagen in Amsterdam) is het totale aantal woningen in dit blok dus 152. Om dit aantal woningen te kunnen afleiden van de huizenblokkenkaart is het nodig om de binnenruimte van de huizenblokken af te trekken. Dit kan gedaan worden door van een gemiddelde diepte van de huizen uit te gaan en deze dieptemaat te gebruiken om inwendige buffers van alle blokken te maken. Wij zijn uitgegaan van een gemiddelde dieptemaat van 12 meter (in Amsterdam zijn de meeste woningen dwars op de richting van de straat gebouwd). Indien In figuur 8 is het resultaat te zien van de bufferoperatie op de huizenblokken met een inwendige bufferoperatie.



*Figuur II.9 Inwendige buffer 12 meter van huizenblok in Amsterdam Oud Zuid.*

Nu is van elke buffer afzonderlijk of van alle buffers tezamen de oppervlakte te bepalen die gedeeld kan worden door een gemiddeld woonareaal. De bufferzone van het betreffende blok in Amsterdam Oud-Zuid blijkt een grootte te hebben van 3160 $m^2$ (footprint). Indien uitgegaan wordt van een woonareaal van 85 $m^2$ per pand (een redelijke maat voor dit gebied), betekent dit een aantal van 37 huizen, maal 4 bouwlagen = 148 huizen. Dit aantal komt goed overeen met het aantal werkelijke huizen dat 152 bedraagt voor dit blok.

De vraag is nu echter of deze benadering ook opgaat voor andere wijken en andere steden in Nederland. Hiervoor is het belangrijk dat uitgegaan kan worden van een gemiddeld aantal van 4 woonlagen, van een gemiddelde huizendiepte en een gemiddelde oppervlakte per woning. De gemiddelde huizendiepte zal afhangen van de wijze waarop de woning is gepositioneerd ten opzichte van de weg, in dwars of lengterichting. Indicaties voor het gemiddelde woningoppervlak in Nederland worden gegeven door het Ministerie van VROM (2003). De Nederlandse woning heeft een gemiddeld gebruiksoppervlak (GBO) van 104 $m^2$. Bij eengezinswoningen is dat 118 m2, bij meergezinswoningen 72 $m^2$. Deze getallen moeten vertaald worden naar bruto vloeroppervlakte (BVO) om ze te kunnen vergelijken met de door ons bepaalde woningoppervlakten. Een indicatie hiervoor wordt gegeven door de berekeningswijze van de GBO en de BVO, geïllustreerd met plattegronden van appartementen in de NEN publicatie (1997) 'Oppervlakten en inhouden van gebouwen'. Hieruit blijkt dat de GBO circa 85% bedraagt van de BVO. Dat wil zeggen dat in Nederland grofweg uitgegaan kan worden van een gemiddeld BVO van 120 $m^2$ (respectievelijk 135 $m^2$ voor eengezinswoningen en 83 $m^2$ voor meergezinswoningen). Aangezien de huizenblokken kaart vooral betrekking heeft op appartementcomplexen in de stedelijke gebieden, lijkt onze aanname van 85 $m^2$ per wooneenheid in een meergezinswoning dus redelijk.

Vervolgens zijn 10 intersecties uitgevoerd tussen respectievelijk de verschillende buffers van 1 tot en met 5 km en de huizenblokkenkaart en de 5 bufferzones met de huizenkaart.

*Huizen (vrijstaand en rijtjeshuizen) binnen bufferzones*

In Tabel II.5 zijn alle polygonen weergegeven van de huizenkaart die binnen de gedefinieerde bufferzones vallen.

*Tabel II.5 Oppervlaktes invloedsgebieden AVIs en aantal polygonen.*

| Invloedsgebied rond AVI | Oppervlakte in km2 | Aantal polygonen |
|---|---|---|
| 1000 meter | 35,2 | 530 |
| 2000 meter | 139,2 | 5858 |
| 3000 meter | 311,9 | 22418 |
| 4000 meter | 553,4 | 44970 |
| 5000 meter | 863,7 | 68111 |

Van deze polygonenkaarten zijn het aantal woningen per invloedsgebied als volgt afgeleid en samengevat in tabel 6.

Van de 530 polygonen in het 1000 meter invloedsgebied zijn er 426 > 200 m2. Deze hebben een gezamenlijke oppervlakte van 322.791 m2 / 150 = 2.152 huizen. Als we hier het aantal van 102 huizen < 200 m2 bij optellen komen we uit op 2.254 huizen. De huizendichtheid komt dan uit op 64 huizen per km2.

Van de 5.858 polygonen in het 2000 meter invloedsgebied zijn er 3.612 > 200 $m^2$. Deze hebben een gezamenlijke oppervlakte van 2.042.799 $m^2$ / 150 = 13.619 huizen. Als we hier het aantal van 2.244 huizen < 200 $m^2$ bij optellen komen we uit op 15.863 huizen. De huizendichtheid komt dan uit op 114 huizen per $km^2$.

Van de 22.418 polygonen in het 3000 meter invloedsgebied zijn er 11.665 > 200 $m^2$. Deze hebben een gezamenlijke oppervlakte van 6.178.804 $m^2$ / 150 = 41.192 huizen. Als we hier het aantal van 10.751 huizen < 200 $m^2$ bij optellen komen we uit op 51.943 huizen. De huizendichtheid komt dan uit op 167 huizen per $km^2$.

Van de 44.970 polygonen in het 4000 meter invloedsgebied zijn er 21.855 > 200 $m^2$. Deze hebben een gezamenlijke oppervlakte van 11.442.729 $m^2$ / 150 = 76.285 huizen. Als we hier het aantal van 23113 huizen < 200 m2 bij optellen komen we uit op 99.398 huizen. De huizendichtheid komt dan uit op 180 huizen per $km^2$.

Van de 68.111 polygonen in het 5000 meter invloedsgebied zijn er 34.047 > 200 $m^2$. Deze hebben een gezamenlijke oppervlakte van 16.765.323 $m^2$ / 150 = 111.769 huizen. Als we hier het aantal van 34.064 huizen < 200 m2 bij optellen komen we uit op 145.833 huizen. De huizendichtheid komt dan uit op 169 huizen per $km^2$.

*Table II.6 Oppervlaktes invloedsgebieden AVIs en gecorrigeerd aantal huizen.*

| Invloedsgebied rond AVI | Oppervlakte in $km^2$ | Aantal huizen |
|---|---|---|
| 1000 meter | 35,2 | 2.254 |
| 2000 meter | 139,2 | 15.863 |
| 3000 meter | 311,9 | 51.943 |
| 4000 meter | 553,4 | 99.398 |
| 5000 meter | 863,7 | 145.833 |

*Overige huizen (in huizenblokken) binnen bufferzones*

In Tabel II.7 zijn alle polygonen weergegeven van de gecorrigeerde huizenblokkaarten die binnen de gedefinieerde bufferzones vallen.

*Tabel II.7 Oppervlaktes invloedsgebieden AVIs en aantal polygonen.*

| Invloedsgebied rond AVI | Oppervlakte in $km^2$ | Aantal polygonen |
|---|---|---|
| 1000 meter | 35,2 | 79 |
| 2000 meter | 139,2 | 220 |
| 3000 meter | 311,9 | 684 |
| 4000 meter | 553,4 | 1.023 |
| 5000 meter | 863,7 | 1.511 |

Van deze polygonenkaarten zijn het aantal woningen per invloedsgebied als volgt afgeleid en samengevat in tabel 8.

De 79 polygonen in het 1000 meter invloedsgebied hebben een gezamenlijke oppervlakte van 273.757 $m^2$. Als deze oppervlakte gedeeld wordt door een gemiddelde BVO van 85 $m^2$ en vermenigvuldigd wordt met een gemiddelde van 4 woonlagen, komt het totale aantal woningen in de huizenblokken uit op 12.882.

De 220 polygonen in het 2000 meter invloedsgebied hebben een gezamenlijke oppervlakte van 736.756 $m^2$. Als deze oppervlakte gedeeld wordt door een gemiddelde BVO van 85 $m^2$ en vermenigvuldigd wordt met een gemiddelde van 4 woonlagen, komt het totale aantal woningen in de huizenblokken uit op 34.671.

De 684 polygonen in het 3000 meter invloedsgebied hebben een gezamenlijke oppervlakte van 2.264.008 $m^2$. Als deze oppervlakte gedeeld wordt door een gemiddelde BVO van 85 $m^2$ en vermenigvuldigd wordt met een gemiddelde van 4 woonlagen, komt het totale aantal woningen in de huizenblokken uit op 106.542.

De 1023 polygonen in het 4000 meter invloedsgebied hebben een gezamenlijke oppervlakte van 3.427.962 $m^2$. Als deze oppervlakte gedeeld wordt door een gemiddelde BVO van 85 $m^2$ en vermenigvuldigd wordt met een gemiddelde van 4 woonlagen, komt het totale aantal woningen in de huizenblokken uit op 161.316.

De 1511 polygonen in het 5000 meter invloedsgebied hebben een gezamenlijke oppervlakte van 5.035.818 $m^2$. Als deze oppervlakte gedeeld wordt door een gemiddelde BVO van 85 m2 en vermenigvuldigd wordt met een gemiddelde van 4 woonlagen, komt het totale aantal woningen in de huizenblokken uit op 236.980.

*Tabel II.8 Oppervlaktes invloedsgebieden AVIs en gecorrigeerd aantal huizen.*

| Invloedsgebied rond AVI | Oppervlakte in km$^2$ | Aantal woningen |
|---|---|---|
| 1000 meter | 35,2 | 12.882 |
| 2000 meter | 139,2 | 34.671 |
| 3000 meter | 311,9 | 106.542 |
| 4000 meter | 553,4 | 161.316 |
| 5000 meter | 863,7 | 236.980 |

Door het afgeleide aantal huizen/woningen uit de huizenkaart en huizenblokkenkaart, krijgen we het totale aantal huizen/woningen in de verschillende invloedsgebieden rond de AVI's. Door deze aantallen te delen door de oppervlaktes van de verschillende invloedsgebieden wordt de huizen/woningendichtheid per invloedszone verkregen. Deze cijfers zijn weergegeven in Tabel II.9.

*Tabel II.9 Oppervlaktes invloedsgebieden AVIs, totale aantal huizen en huizen/woningendichtheid.*

| Invloedsgebied rond AVI | Oppervl. in km$^2$ | Aantal huizen | Aantal blok-woningen | Totaal aantal huizen/woningen | Dichtheid (aantal/km$^2$) |
|---|---|---|---|---|---|
| 1000 meter | 35,2 | 2.254 | 12.882 | 15.136 | 430 |
| 2000 meter | 139,2 | 15.863 | 34.671 | 50.534 | 363 |
| 3000 meter | 311,9 | 51.943 | 106.542 | 158.485 | 508 |
| 4000 meter | 553,4 | 99.398 | 161.316 | 260.714 | 471 |
| 5000 meter | 863,7 | 145.833 | 236.980 | 382.813 | 443 |

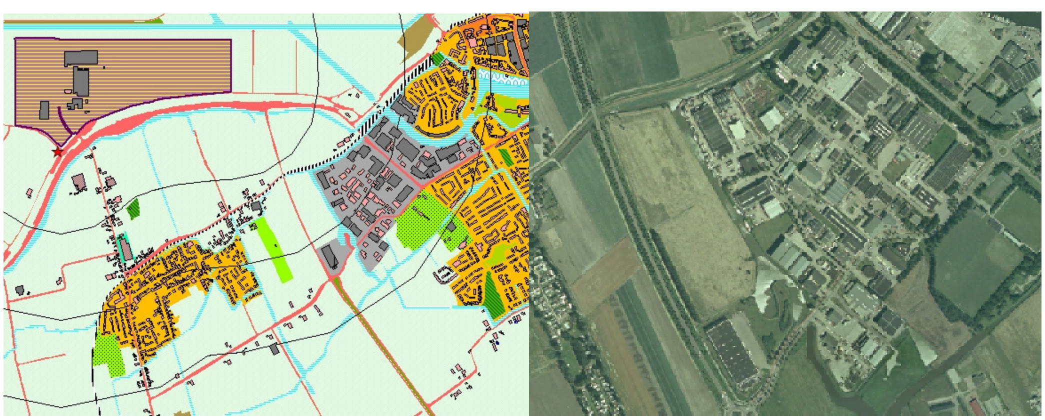
Uit de cijfers in Tabel II.9 valt op dat de dichtheid van huizen/woningen hoger is in een 1000 meter zone rond de AVI's dan in een 2000 meter zone rond AVI's. Dit is tegen de verwachting in, maar kan verklaard worden als gekeken wordt naar de huizenspreiding rondom de verschillende AVI's. Rondom bijna alle AVI's zijn weinig tot geen huizen en woningen te vinden in een straal van 1000 meter rondom de AVI's op één uitzondering na: Afvalverwerking Rotterdam aan de Brielselaan 175, te Rotterdam. Rondom deze AVI bevinden zich 1.445 huizen + 12.736 woningen in huizenblokken = 14.181 woningen/huizen. Dit betreft ca. 94% van alle huizen/woningen in een straal van 1000 meter rondom AVI's in Nederland. Voor de overige invloedszones van 2000 tot 5000 meter neemt de huizen/woningendichtheid volgens verwachting eerst toe om daarna weer af te nemen. De hoogste huizen/woningendichtheid wordt gevonden in een invloedsgebied van 3000 meter rondom de AVI's.

### Beperkingen in de analyse

- Classificatie top10 vector huizen / bebouwd gebied. Top10 vector bevat een aparte huizenlaag, maar deze laag is verre van compleet als deze wordt vergeleken met het bebouwde gebied in de vlakkenlaag en met luchtfoto's van de zelfde gebieden. Dit probleem lijkt vooral in de stedelijke gebieden te spelen. Hieronder in figuur 9 is dat goed te zien voor het gebied rond de AVR in Rotterdam aan de Brielseweg. Het is onduidelijk waarom sommige huizen wel apart (in rood) en sommige alleen als huizenblok (in grijs) in de vlakkenkaart worden weergegeven. Probleem met de weergave in huizenblokken is dat hier nog moeilijker het aantal huizen bepaald kan worden, daar het blok ook de binnenruimtes omvat. Figuur 10 laat zien dat in het landelijke gebied de bebouwing uit de vlakkenkaart hoofdzakelijk uit kantoren en fabriekshallen bestaat;

*Figuur II.10 Vergelijking top10vector met luchtfoto 2003 voor stedelijk gebied rond Brielseweg in Rotterdam.*

- Een ander probleem met de gebouwen op de vlakkenkaart is dat deze zowel woonhuizen als andere gebouwen zoals kantoren en fabrieken weergeeft. In de landelijke gebieden is dit geen probleem, omdat hier de huizenkaart vrijwel alleen huizen weergeeft en de gebouwen op de vlakkenkaart andersoortige gebouwen zijn. In het havengebied zoals in Figuur II.10 bijvoorbeeld, is dit wel een probleem want hier worden alle industriële gebouwen meegeteld indien deze kaart voor de analyse gebruikt wordt. Dit probleem is echter opgelost door alleen die polygonen te selecteren die binnen bepaalde klassen van de bodemstatistiek kaart vallen, zie figuren II.7a en II.7b;
- Een mogelijkheid om de uitgevoerde analyses en de uitgangspunten te toetsen zou een controle zijn voor bijvoorbeeld 2 of 3 gebieden (bijv. een landelijk gebied, een industriegebied en een woongebied bij een grotere stad) waarbij een precieze telling wordt uitgevoerd van het aantal woningen aan de hand van een grootschalige gemeentekaart (bijv. 1:1000).



*Figuur II.11 Vergelijking top10vector met luchtfoto 2003 voor landelijk gebied rond stortplaats Wieringermeer.*

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## Bijlage 1: Actieve stortplaatsen met xy-coordinaten

| Gemeente | Plaats | Adres | Postcode | Provincie | Woningen | Oppervlakte | Gemiddeld | Gewogen gem. | latitude | longitude | adresbron |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Groningen | | Winschoterweg 1 | 9723 CG | Groningen | 82691 | 83.69 | 988.06 | 20.58 | 53.19684 | 6.62267 | MP |
| Skarsterlan | Oudehaske | De Dolten 11 | 8465 SB | Friesland | 10848 | 216.89 | 50.02 | 2.70 | 52.96866 | 5.86935 | TG |
| Midden-Drenthe | Wijster | VAMweg 7 | 9418 TM | Drenthe | 12730 | 345.82 | 36.81 | 3.17 | 52.79291 | 6.51528 | TG |
| Assen | Ubbena | Vriezerhoek 1 | 9492 TH | Drenthe | 25951 | 83.48 | 310.86 | 6.46 | 53.05486 | 6.57082 | M24 |
| Hengelo | | Boldershoekweg 51 | 7554 RT | Overijssel | 35389 | 61.78 | 572.82 | 8.81 | 52.23893 | 6.7846 | TG |
| Hardenberg | Rheezerveen | Ommerweg 69 | 7797 RC | Overijssel | 20767 | 317.24 | 65.46 | 5.17 | 52.55216 | 6.52725 | TG |
| Lochem | | Hagendijk 1 | 7241 NR | Gelderland | 7584 | 128.8 | 58.88 | 1.89 | 52.17018 | 6.44846 | TG |
| Beuningen | Weurt | Nieuwe Pieckelaan 1 | 6551 DX | Gelderland | 9595 | 47.17 | 203.41 | 2.39 | 51.85067 | 5.78398 | TG |
| Geldermalsen | | Meersteeg 15 | 4191 NK | Gelderland | 9637 | 101.67 | 94.79 | 2.40 | 51.86888 | 5.3211 | TG |
| Voorst | Wilp | Sluinerweg 12 | 7384 SC | Gelderland | 8614 | 126.52 | 68.08 | 2.14 | 52.20147 | 6.07807 | TG |
| Ermelo | | Jhr Dr C. Sandbergweg 115 | 3852 PT | Gelderland | 8982 | 87.38 | 102.79 | 2.24 | 52.30755 | 5.68252 | TG |
| Barneveld | | Wencopperweg 33 | 3771 PN | Gelderland | 16481 | 176.74 | 93.25 | 4.10 | 52.16441 | 5.62145 | M24 |
| Zevenaar | | Doesburgseweg 16d | 6902 PN | Gelderland | 11052 | 27.65 | 399.71 | 2.75 | 51.94339 | 6.08368 | TG |
| Almere | | Kemphaanweg 2 | 1358 AB | Flevoland | 63771 | 248.77 | 256.35 | 15.87 | 52.33856 | 5.2686 | TG |
| Amersfoort | Hoogland | Lindeboomseweg 15 | 3828 NG | Utrecht | 54031 | 63.78 | 847.15 | 13.45 | 52.20295 | 5.39051 | MP |
| Alkmaar | | Boekelerdijk 13a | 1812 LV | Noord-Holland | 41008 | 31.22 | 1313.52 | 10.21 | 52.60508 | 4.76343 | M24 |
| Zaanstad | Assendelft | Nauerna 1 | 1566 PB | Noord-Holland | 58512 | 83.04 | 704.62 | 14.56 | 52.44112 | 4.74601 | MP |
| Wieringermeer | Middenmeer | Koggenrandweg 1 | 1775 RG | Noord-Holland | 4992 | 307.76 | 16.22 | 1.24 | 52.76662 | 5.06721 | kaart website |
| Dordrecht | | Baanhoekweg 20 | 3313 LA | Zuid-Holland | 52338 | 99.45 | 526.27 | 13.03 | 51.81627 | 4.71592 | MP |
| Dordrecht | | Baanhoekweg 92a | 3313 LP | Zuid-Holland | 52338 | 99.45 | 526.27 | 13.03 | 51.81513 | 4.74931 | M24 onduidelijk |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Rotterdam | Maasvlakte Rotterdam | Loswalweg 50 | 3199 LG | Zuid-Holland | 286762 | 304.24 | 942.55 | 71.37 | 51.92604 | 4.02849 | M24 |
| Terneuzen | Sluiskil | Koegorsstraat 19 | 4541 HT | Zeeland | 24872 | 318.78 | 78.02 | 6.19 | 51.27355 | 3.87029 | M24 onduidelijk |
| Borsele | Nieuwdorp | Frankrijkweg 2 | 4455 TR | Zeeland | 8831 | 194.44 | 45.42 | 2.20 | 51.45568 | 3.70879 | MP |
| Tilburg | | Vloeiveldweg 8 | 5048 TD | Noord-Brabant | 82757 | 118.83 | 696.43 | 20.60 | 51.60072 | 5.06004 | TG |
| Bergen op Zoom | | Moervaart 25 | 4622 RR | Noord-Brabant | 27770 | 93.13 | 298.19 | 6.91 | 51.50527 | 4.33918 | TG |
| Nuenen c.a. | | Gulberg 9 | 5674 TE | Noord-Brabant | 9081 | 34.11 | 266.23 | 2.26 | 51.45024 | 5.56845 | MP |
| Schijndel | | Vlagheide 10 | 5482 NM | Noord-Brabant | 8740 | 41.65 | 209.84 | 2.18 | 51.60148 | 5.48416 | TG |
| Landgraaf | | Europaweg Noord 179 | 6374 CH | Limburg | 17469 | 24.69 | 707.53 | 4.35 | 50.92819 | 6.02725 | TG |
| Ambt Montfoort | | Maasbrachterweg 3 | 6065 NN | Limburg | 4317 | 44.28 | 97.49 | 1.07 | 51.14221 | 5.934 | M24 onduidelijk |
| Weert | | Hazenweg 1 | 6006 TC | Limburg | 20115 | 105.44 | 190.77 | 5.01 | 51.21498 | 5.65052 | M24 |
| | | | | | 4017.89 | 358.93 | 268.31 | | | | |

MP = Microsoft MapPoint

TG = kaartservice Nationale TelefoonGids (http://www.nationaletelefoongids.nl)

M24 = Map24 (http://www.nl.map24.com/)